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LEACHATE TESTING OF HAMLET CITY LAKE
NORTH CAROLINA, SEDIMENT

by

James M. Brannon, Tommy E. Myers, Cynthia B. Price

Environmental Laboratory

DEPARTMENT OF THE ARMY

Waterways Experiment Station, Corps of Engineers
3909 Halls Ferry Road, Vicksburg, Mississippi 39180-6199



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13. ABSTRACT (Maximum 200 words) Sediment leaching studies of Hamlet City Lake, Hamlet, NC, were conducted in laboratories at the U.S. Army Engineer Waterways Experiment Station. The purpose of these studies was to provide quantitative information on the potential for leachate impacts on groundwaters if dredged material from Hamlet City Lake were placed in a confined disposal facility (CDF) or under disposal conditions similar to landfarming. The study involved three elements: batch leach tests, column leach tests, and simulations using the Hydrologic Evaluation of Landfill Performance (HELP) model. Batch tests were conducted to determine intrinsic leaching characteristics of solids in Hamlet City Lake sediment. Column tests were conducted as a physical analog of continuous flow leaching in a CDF. HELP model simulations were conducted to simulate the generation of leachate by infiltration and to evaluate the relative effectiveness of a disposal site liner. (Continued)				
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Results of this study showed that, under disposal conditions similar to landfarming, organic contaminants such as polycyclic aromatic hydrocarbons (PAHs) and total recoverable petroleum hydrocarbons (TRPHs) will decrease in concentration as the result of volatilization and/or biodegradation. Neither PAHs or TRPHs should, therefore, pose serious environmental problems. The data collected in this study are insufficient to estimate the rate of disappearance under field conditions; however, under selected laboratory conditions, more than 80 percent of the TRPHs disappeared in 6 months.

The pH of Hamlet City Lake sediment was lower (<6.0) than usual while the sediment was anaerobic, and was further decreased to less than 4.0 after oxidation. The data on metals leaching under disposal conditions similar to landfarming indicated that leachate quality was not seriously aggravated by dewatering and aeration of the sediment. Of the seven metals investigated, three (arsenic, copper, and lead) were more mobile under anaerobic leaching conditions than under aerobic leaching conditions. Mercury did not leach under anaerobic or aerobic leaching conditions. For the three metals that were more mobile under aerobic leaching conditions (cadmium, nickel, and zinc), the mass release under aerobic batch leaching conditions was higher than that under anaerobic batch leaching conditions by a factor of approximately 2.5. A factor of only 2.5 does not indicate substantial mobilization relative to the anaerobic condition.

Comparisons of maximum contaminant concentrations in anaerobic leachates to drinking water limits and water quality criteria indicated that both Federal drinking water criteria and State of North Carolina groundwater standards could be exceeded for arsenic and lead. State groundwater standards for cadmium and nickel could also be exceeded. Anaerobic leaching conditions are representative of conditions in a CDF. Aerobic conditions are representative of disposal conditions similar to landfarming. Federal fresh water quality criteria and State of North Carolina surface water standards could be exceeded for cadmium and zinc by leachate from aerobic sediment.

Six disposal scenarios were simulated using the HELP model. The simulations indicated annual leachate flow of 2,125 to 4.2 million cu m, depending on disposal area design. Placement of a 15-cm lift of dredged material on highly permeable soils with no leachate controls produced the largest flow, and confined disposal of a 3-m lift with a composite liner produced the smallest flow.

Low sediment pH following oxidation may inhibit plant growth if the dredged material is spread over fields. Sediment conditioning to increase pH and create more hospitable conditions for plant growth will be needed if this material is to be placed on open fields. Metal concentrations in surface runoff from the conditioned material should be evaluated to determine if surface water impacts are likely. Groundwater impacts of anaerobic sediment are not a concern during disposal similar to landfarming because the sediment is rapidly oxidized. Potential impacts should be evaluated under aerobic sediment conditions. Metal transport to surface waters poses the main problem because PAHs are present in low concentrations and both PAHs and TRPHs are rapidly lost by volatilization and biodegradation from the sediment during exposure to the air.

PREFACE

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This report was prepared by Dr. James M. Brannon, Aquatic Processes and Effects Group (APEG), Ecosystem Research and Simulation Division (ERSD), EL; Mr. Tommy E. Myers, Water Supply and Waste Treatment Group (WSWTG), Environmental Engineering Division (EED), EL; and Ms. Cynthia B. Price, also of the APEG. Dr. Judith C. Pennington, APEG, and Dr. Christian McGrath, Water Quality Modeling Group, ERSD, EL, were technical reviewers for this report. Ms. Tiarra A. McDonald, ASCL Corporation, and Mr. Antoine Bargins, WSWTG, provided laboratory support and data tabulation. Ms. Karen Myers, Mr. Don Brown, Mr. Charles White, and Ms. Allyson Lynch, all of the Analytical Laboratory Group (ALG), EED, provided analytical support under the direct supervision of Ms. Ann Strong, Chief, ALG.

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At the time of publication of this report, Director of WES was Dr. Robert W. Whalin. Commander was COL Leonard G. Hassell, EN.

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CONVERSION FACTORS, NON-SI TO SI
UNITS OF MEASUREMENT

Non-SI units of measurement used in this report can be converted to SI units as follows:

<u>Multiply</u>	<u>By</u>	<u>To Obtain</u>
cubic feet	0.02831685	cubic meters
cubic yards	0.7645549	cubic meters
feet	0.3048	meters
gallons (U.S. liquid)	3.785412	liters
gallons (U.S. liquid) per acre	0.00093	cubic liters per square meter
inches	2.54	centimeters
pounds (force) per square inch	6.894757	kilopascals

LEACHATE TESTING OF HAMLET CITY LAKE,
NORTH CAROLINA. SEDIMENT

PART I: INTRODUCTION

1. When dredged material is placed in a confined disposal facility (CDF), contaminants may be mobilized and transported to the site boundaries by leachate and seepage. Subsurface drainage and seepage through foundation soils and dikes may then reach adjacent surface and ground waters and act as a source of contamination. Leachate quality predictions for CDFs can be used to establish sound engineering and environmental data for the selection, design, and management of CDFs.

2. Hamlet City Lake in Hamlet, NC, has been included in the Federal Lake Cleanup Program authorized by the Water Resources Development Act of 1986. The authorizing document instructs the U.S. Army Corps of Engineers to carry out a program for removal of the approximately 230,000 cu yd* of material from the lake which has constrained the town's plans for use of the lake. Disposal of the material within an upland CDF or as a layer on open fields is being considered. To assist in developing plans for the dredging and disposal of material from Hamlet City Lake, the U.S. Army Engineer District, Wilmington, is using the Long-Term Management Strategy (Francingues et al. 1985) that is incorporated by reference in the Corps' dredging regulation (33 CFR Parts 209, 335-339), of which leachate testing is a part. This report presents an evaluation of potential leachate quality from Hamlet City Lake sediments. Leachate quality under both anaerobic and aerobic sediment conditions for a number of disposal scenarios was investigated.

* A table of factors for converting non-SI units of measurement to SI units is presented on page 5.

PART II: METHODS AND MATERIALS

Sample Collection

3. Sediment samples from Hamlet City Lake, Hamlet, NC, were obtained on 5 December 1990 by Dan Small, Phil Payonk, and Jenny Owens of the Wilmington District. Sediments were collected from the eight locations indicated in Figure 1 to obtain samples representative of the area to be dredged. Samples were collected using a 10-cm-diam polyvinyl chloride hand corer. The corer was thrust into the sediment until refusal, sealed with a rubber stopper, and retrieved. This process was repeated until 2.5 gal of sediment was collected at each sampling site.

4. Samples were placed into precleaned, high-density polyethylene, 5-gal buckets provided by the WES. Samples were packed in ice and shipped by Federal Express to the WES. Upon arrival at the WES, the eight samples were mixed together to form one composite sample that was used in the leachate tests.

Oxidized Sediment Preparation

5. The sediment used in aerobic testing was first placed into 38-L glass aquariums to a depth of approximately 6 cm. The sediment was allowed to oxidize at ambient temperature. Each week the sediment was thoroughly mixed to expose fresh sediment surfaces to the air. When necessary, deoxygenated distilled-deionized (DDI) water was added to the sediment to maintain the original moisture condition. After 6 months, the sediment was again thoroughly mixed before being used in testing.

Kinetic Batch Testing

6. Batch testing was performed to determine the shaking time necessary to achieve equilibrium or steady-state conditions for metals and polycyclic aromatic hydrocarbons (PAH) leachate concentrations. The general experimental sequence is presented in Table 1.

7. For testing metal releases from anaerobic Hamlet City Lake sediment, triplicate 250-ml polycarbonate centrifuge tubes fitted with a leakproof, airtight top were loaded with 200 g of sediment and deoxygenated DDI water at

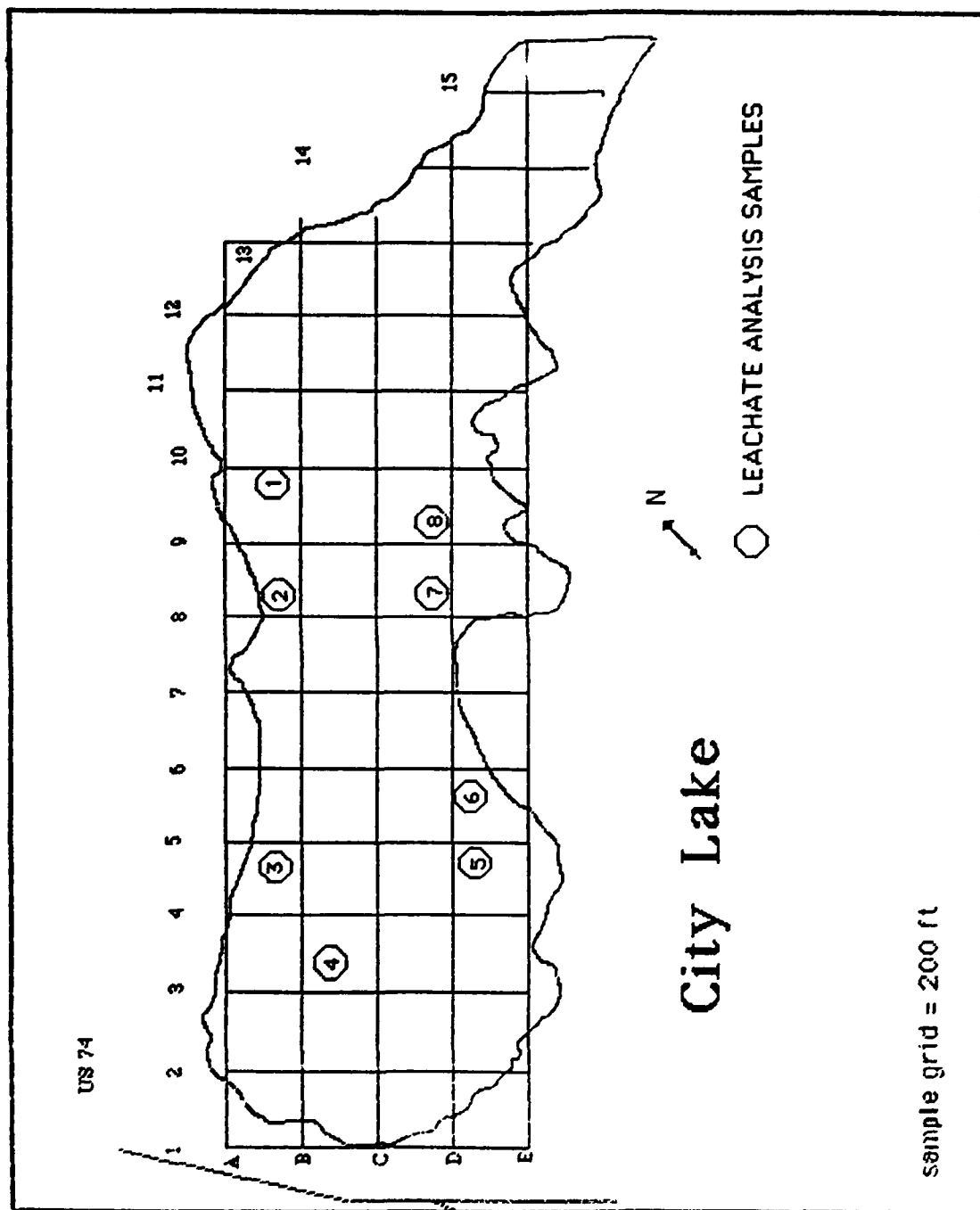


Figure 1. Sediment sampling sites for leachate analysis in Hamlet City Lake

Table 1

Experimental Sequence for Determining Appropriate Shaking Times
for Anaerobic Kinetic Testing of Hamlet City Lake Sediment

<u>Step</u>	<u>Procedure</u>
1	Place sediment in appropriate centrifuge tube (stainless steel for PAHs, or polycarbonate for metals). Add sufficient deoxygenated DDI water to maintain water-to-sediment ratio of 4:1.
2	Place centrifuge tubes for metals on reciprocating shaker and shake at 160 cycles per minute. Place centrifuge tubes for PAHs in rotary mixer and turn at 40 revolutions per minute.
3	Remove tubes (enough for triplicate samples) from shaker at appropriate intervals: 4, 12, 24 hr and 7 days for PAHs, and 4, 12, 24 hr and 2, 4, 7 days for metals.
4	Centrifuge for 30 min at $6,500 \times g$ for PAHs and $13,000 \times g$ for metals.
5	Filter centrifuged leachate through a $0.45\text{-}\mu\text{m}$ pore size membrane filter for metals, and through a Whatman GF/D glass-fiber pre-filter and a Gelman AE glass-fiber filter with a $1\text{-}\mu\text{m}$ nominal pore size for PAHs.
6	Acidify leachate for metals and PAH analysis with Ultrex nitric acid. Store samples for analysis of PAHs in glass bottles at $4\text{ }^{\circ}\text{C}$ until analysis. Store leachate for metals analysis in plastic bottles.

a 4:1 water-to-sediment dry weight ratio. All operations were conducted in a glove box under a nitrogen atmosphere. Nineteen centrifuge tubes were loaded to allow triplicate sampling at 6 hr and at 1, 2, 3, 7, and 10 days, along with a procedure blank. Samples were placed horizontally on a reciprocating shaker at 160 strokes per minute. At designated sampling times, three tubes were removed from the shaker and centrifuged at 9,000 rpm ($13,000 \times g$) for 30 min. The supernatant was filtered under a nitrogen atmosphere through 0.45- μ m membrane filters. The filtrate was acidified to pH 1 with concentrated Ultrex nitric acid and stored in plastic bottles until analyzed.

8. Kinetic testing for PAHs in anaerobic Hamlet City Lake sediment was conducted using specially fabricated 450-ml stainless steel centrifuge tubes. Twenty-one centrifuge tubes (sufficient for three replicates), double-rinsed with acetone and dried, were loaded with sufficient sediment and deoxygenated DDI water to obtain a 4:1 water-to-sediment dry weight ratio. The total mass (approximately 350 g) of sediment and water was adjusted to allow the tube to be safely centrifuged at 6,200 rpm ($6,500 \times g$). All operations were conducted under a nitrogen atmosphere. The tubes were placed in a rotary tumbler and turned end over end at 40 rpm for periods of 1, 2, and 7 days. At the appointed times, the samples were removed from the tumbler and centrifuged for 30 min. Resulting supernatants were filtered through a Whatman GF/D prefilter and a Gelman AE filter with a nominal pore size of 1.0 μ m. The filters (Whatman GF/D prefilters and Gelman AE filters) are binderless glass-fiber and contain no detectable organic contaminants. As a further precaution against contamination, the filters used in this study were combusted at 400 °C prior to use. Filtration was conducted under a nitrogen atmosphere; samples for PAHs were acidified with Ultrex nitric acid and stored in acetone-rinsed 2-L glass bottles at 4 °C until analyzed.

Sequential Batch Testing

9. A 4:1 water-to-sediment ratio and a shaking time of 24 hr were used in the sequential batch leach tests for Hamlet City Lake sediment. General test procedures for assessing steady-state leachate and sediment metal and PAH concentrations are detailed in Table 2. Sequential batch tests for metals under anaerobic conditions were conducted in triplicate 250-ml polycarbonate centrifuge bottles with leakproof caps. Each centrifuge tube was loaded under a nitrogen atmosphere with anaerobic Hamlet City Lake sediment and

Table 2

Test Sequence for Sequential Batch Leaching of Anaerobic Hamlet City Lake
Sediment for Metals, Polycyclic Aromatic Hydrocarbons, and
Total Recoverable Petroleum Hydrocarbons (TRPHs)

<u>Step</u>	<u>Procedure</u>
1	Load sediment into appropriate centrifuge tubes--250-ml polycarbonate for metals and 450-ml stainless steel for TRPHs and PAHs. Add DDI water to each tube to bring final water-to-sediment ratio to 4:1. Load sufficient number of stainless steel tubes to ensure enough leachate for triplicate analysis.
2	Agitate sample, as described in Step 2 of Table 1. Samples for TRPHs are treated the same as samples for PAHs.
3	Centrifuge for 30 min at $6,500 \times g$ for TRPHs and PAHs and $13,000 \times g$ for metals.
4	Filter leachate through $0.45\text{-}\mu\text{m}$ membrane filters for metals or through Whatman GD/F glass-fiber prefilters followed by Gelman AE glass-fiber filters of $1.0\text{-}\mu\text{m}$ nominal pore size for TRPHs and PAHs. Set aside a small amount of leachate prior to acidification for analysis of pH, conductivity, and total organic carbon.
5	Return to Step 2 after replacing leachate with deoxygenated DDI water. Repeat the entire procedure the desired number of times.

Notes: Testing sequence is the same for aerobic sediments except that anaerobic integrity is not maintained.

deoxygenated DDI water to a 4:1 water-to-sediment ratio. Tubes were mechanically shaken for 24 hr and centrifuged at $13,000 \times g$ for 30 min. Most of the supernate from each 250-ml centrifuge bottle was filtered through a $0.45\text{-}\mu\text{m}$ membrane filter. An unfiltered aliquot was analyzed for pH using a combination electrode and a millivolt meter and for conductivity using a Yellow Springs Instrument Company conductivity meter and cell. Fresh deoxygenated DDI water was added to replace the leachate removed for analysis. The procedure described above for sequentially contacting anaerobic Hamlet City Lake sediment with clean water was repeated seven times. The same general procedure was repeated for aerobic batch leach tests for metals, except that anaerobic conditions were not maintained.

10. Testing of Hamlet City Lake sediment for TRPHs and PAHs was conducted as described for metals except that 450-ml stainless steel centrifuge tubes were used. Filtration procedures used for TRPHs and PAHs were as previously described for kinetic batch testing. An aliquot of filtered leachate was set aside from both the anaerobic and aerobic tests for analysis of total organic carbon (TOC). After each cycle, the sediment was remixed with DDI water, shaken for 24 hr, and then processed as previously described. Three replicates were taken through six leach cycles for the anaerobic and aerobic testing.

Column Leach Testing

11. Column leach tests were conducted in large-diameter packed columns (Figure 2) designed specifically for evaluation of fine-grained sediment and dredged material. The apparatus was designed to simulate leaching of dredged material in a continuous-flow mode, minimize wall effects, hold pore water velocities to about 10^{-5} cm/sec or less, and produce sufficient sample volume for chemical analysis of fractional pore volume samples. The oxygen demand of dredged material is sufficient to maintain anaerobic conditions in dredged material leached in the column. Column design is described by Myers, Gambrell, and Tittlebaum (1991).

12. Sediment was placed in leaching columns at in situ water content in small increments, and manually vibrated with a spatula to minimize air entrapment. Since the water content of the sediment was high, the sediment was easily worked using manual vibration techniques. Water content and specific gravity of the sediment were determined according to methods described in

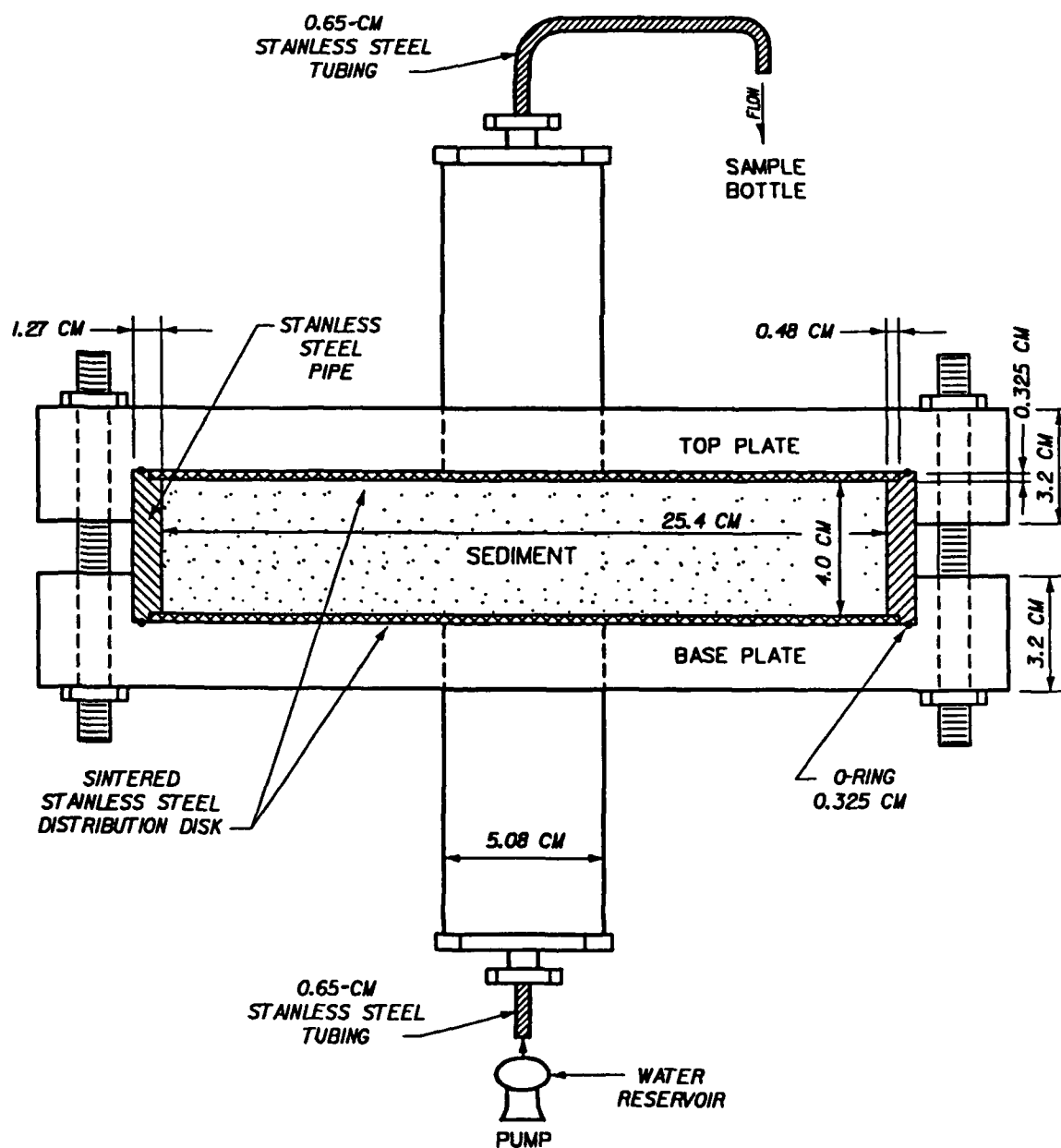


Figure 2. Dredged material leaching column

U.S. Army Corps of Engineers (1970). These data were used to calculate sediment porosity. After the columns were filled, DDI water was pumped in upflow mode through the columns with a constant volume discharge pump.

13. Column leach tests were conducted in triplicate for metals and PAHs. Separate column leach tests were run for metals and organic contaminant analysis because of the differences in sample volumes needed to conduct metal (approximately 100 ml) and organic (approximately 1,000 ml) analyses and because of the different preservation techniques required. For each column leachate sample collected, the volume of leachate and time of collection were recorded. As leachate from columns conducted for metals was collected, pH was adjusted to <2 using Ultrex nitric acid. Samples collected for metal analysis were split for metal and TOC analysis. Leachates for PAH analysis were collected under hexane in acetone-rinsed, amber glass jars. The use of hexane to minimize volatile losses of PAHs during sample collection precluded analysis for TOC or TRPH. Leachates for organic analyses were not pH adjusted or otherwise altered. After each leachate sample for chemical analysis (metals or organics) was collected, an additional sample of 15 to 20 ml was collected for pH and electrical conductivity measurement.

Chemical Analysis

14. Leachate and sediment samples were analyzed by the Analytical Laboratory Group of the Environmental Laboratory for TRPHs, PAHs (naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz(a)anthracene, chrysene, benzo(b) + benzo(k)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, dibenz(a,h)anthracene, benzo(ghi)perylene), and metals (nickel (Ni), cadmium (Cd), lead (Pb), mercury (Hg), arsenic (As), zinc (Zn), copper (Cu)).

15. TRPH was determined using Method 418.1 in Ballinger (1979). PAHs were determined with a Waters High Performance Liquid Chromatograph with a photodiode array detector and a fluorescence detector using Method 8310 (U.S. Environmental Protection Agency, USEPA 1986). All metals except As and Hg were analyzed using directly coupled plasma emission spectroscopy on a Beckman Spectraspan IIIB plasma emission spectrometer or by atomic absorption spectroscopy using a Perkin-Elmer model 5000 atomic absorption spectrometer coupled with a Perkin-Elmer model 500 hot graphite atomizer following appropriate sample digestion procedures (Ballinger 1979). Arsenic in leachate and

sediment samples was determined by hydride generation (Ballinger 1979) using a Perkin-Elmer 305 atomic absorption spectrophotometer coupled with a Perkin-Elmer model MHC-10 hydride generator. Mercury was analyzed by the cold vapor technique (Ballinger 1979). Total organic carbon was analyzed in leachate and sediment samples using an Oceanographic International 543B organic carbon analyzer and standard procedures (Ballinger 1979).

Data and Statistical Analysis

16. All statistical analyses were conducted using methods developed by the Statistical Analysis Systems Institute (Barr et al. 1976). To test for differences between means, analysis of variance procedures were used.

PART III: RESULTS

Sediment Contaminant Concentrations

17. Bulk sediment concentrations of metals, PAHs, and TRPHs in Hamlet City Lake sediments are presented in Table 3. Metals concentrations in the sediment ranged from 0.23 mg/kg for Hg to 216 mg/kg for Pb. Metal concentrations were not reanalyzed following air oxidation because metals are not lost from the sediment as a result of this procedure. Naphthalene, phenanthrene, fluoranthene, pyrene, fluorene, anthracene, and TRPH were the only organic compounds detected in either anaerobic or aerobic sediments. Concentrations of these constituents were higher in anaerobic sediments than in aerobic sediments, a result of degradation or volatilization during the oxidation process. However, concentrations of PAH compounds were low even in anaerobic sediment, with fluoranthene highest at 5.17 mg/kg. Concentration of TRPHs, which constitutes a wide variety of compounds associated with petroleum, was 14,000 mg/kg under anaerobic conditions but dropped to 2,200 mg/kg following aerobic incubation.

18. The USEPA Toxicity Characteristic Leaching Procedure was also conducted upon Hamlet City Lake sediments. Results are summarized in Appendix A.

Kinetic Batch Testing

19. Kinetic batch testing was performed to confirm previous experiments indicating that a 24-hr shaking time was sufficient to achieve steady-state conditions. To test this premise, 1-day concentrations were compared to concentrations obtained at other sampling times during kinetic batch testing. Concentrations of metals at times less than 24 hr were less than or equivalent to the 24-hr values (Table 4). Metal concentrations after 24 hr did not significantly exceed the 24-hr results until day 7, when As, Zn, Cu, and Pb concentrations rose significantly above those at 24 hr. In previous leaching studies with sediments from Indiana Harbor, Indiana (Environmental Laboratory 1987), and Everett Harbor, Washington (Palermo et al. 1989), 1 day was sufficient to attain steady-state or worst-case metal concentrations. Metals in Hamlet City Lake sediment deviated only slightly from this kinetic behavior. Therefore, use of a 24-hr shaking time appeared justified.

Table 3
Contaminant Concentrations (mg/kg) in Anaerobic and Aerobic
Hamlet City Lake Sediment

<u>Parameter</u>	<u>Anaerobic</u>	<u>Aerobic</u>
As	12.4	NT*
Zn	124	NT*
Cu	202	NT*
Ni	12.5	NT*
Pb	216	NT*
Cd	0.58	NT*
Hg	0.23	NT*
Naphthalene	0.438	0.209
Phenanthrene	1.40	0.547
Anthracene	0.151	0.044
Fluoranthene	5.17	3.13
Benz(a)anthracene	<0.000002	<0.000002
Chrysene	<0.00003	<0.00003
Benzo(bjk)fluoranthenes	<0.000003	<0.000003
Benzo(a)pyrene	<0.000004	<0.000004
Indeno(1,2,3-cd)pyrene	<0.000008	<0.000008
Dibenz(a,h)anthracene	<0.00001	<0.00001
Pyrene	3.13	1.97
Fluorene	0.096	0.040
Benzo(ghi)perylene	<0.000013	<0.000013
Acenaphthylene	<0.0004	<0.0004
Acenaphthene	<0.0003	<0.0003
TRPH	14,000	2,200

* Not tested.

20. Concentrations of PAH compounds during kinetic testing in anaerobic Hamlet City Lake sediments are presented in Table 5. Results indicate that 24 hr was sufficient for phenanthrene, pyrene, and fluoranthene, the only PAH compounds detected in the leachate, to reach steady state. Detection limits for PAH compounds in leachate are presented in Table 6. Kinetic testing was not conducted for TRPHs because of the large amount of sediment this procedure would have required.

Sequential Batch Leaching

General leachate quality

21. Leachate pH and electrical conductivity values for anaerobic and aerobic sequential batch leaching tests for Hamlet City Lake sediments are presented in Table 7. Leachate pH from oxidized sediment was consistently

Table 4

Leachate Heavy Metal Concentrations* in Anaerobic Hamlet City Lake
Sediment at Different Sampling Times

Parameter	Sampling Time					
	4 hr	12 hr	24 hr	Day 2	Day 4	Day 7
As	0.054(0.02)	0.003(0.003)	0.049(0.02)	0.041(0.004)	<0.010	0.02(0.002)
Zn	0.061(0.003)	0.058(0.006)	0.063(0.004)	0.06(0.003)	0.02(0.003)	0.161(0.03)
Cu	0.07(0.01)	0.015(0.001)	0.05(0.01)	0.05(0.003)	0.03(0.004)	0.144(0.002)
Ni	0.009(0.001)	0.015(0.006)	0.007(0.0001)	0.004(0.004)	0.004(0.003)	0.02(0.003)
Pb	0.06(0.01)	0.02(0.001)	0.03(0.01)	0.06(0.004)	0.02(0.003)	0.11(0.01)
Cd	0.003(0.001)	0.0002(0.0001)	0.001(0.0004)	0.003(0.0001)	0.002(0.0002)	0.001(0.0001)
Hg	0.0001(0.0001)	<0.0004	<0.0004	<0.0004	<0.0004	0.001(0.0003)

* Expressed in milligrams per liter (standard error in parentheses) (N = 3).

Table 5

PAH Concentrations* in Leachate from Anaerobic Hamlet City Lake
Sediment at Different Sampling Times

Parameter	Sampling Time			
	4 hr	12 hr	24 hr	Day 7
Benzo(b)fluoranthene	0.001 (0.00002)	0.0014 (0.00010)	0.0013 (0.00001)	0.0013 (0.00004)
Phenanthrene	0.00004 (0)	<0.00103	0.00011 (0.00001)	0.00016 (0.00002)
Pyrene	0.00043 (0)	0.00050 (0.00002)	0.00051 (0.00021)	0.00077 (0.00005)
Fluoranthene	0.00053 (0)	0.00064 (0.00003)	0.00091 (0.00008)	0.00116 (0.00010)

* Expressed in milligrams per liter (standard error in parentheses) (N = 3).

Table 6

Detection Limits for PAH Compounds (mg/L)
in Leachate from Hamlet City Lake Sediment

<u>Parameter</u>	<u>Detection Limit</u>
Benzo(bjk)fluoranthenes	<0.000018-0.000026
Benzo(a)pyrene	<0.000023-0.000031
Indeno(1,2,3-cd)pyrene	<0.000044-0.000060
Benzo(ghi)perylene	<0.000077-0.000106
Naphthalene	<0.0019-0.0026
Phenanthrene	<0.00064-0.00095
Fluoranthrene	<0.00021-0.00031
Pyrene	<0.00028-0.00058
Benz(a)anthracene	<0.000013-0.000018
Chrysene	<0.00015-0.00021
Acenaphthylene	<0.0012-0.0034
Acenaphthene	<0.0019-0.0026
Fluorene	<0.00021-0.00031
Anthracene	<0.00064-0.00095
Dibenz(a,h)anthracene	<0.000037-0.00004

Table 7

Leachate pH and Conductivity (millisiemens)
in Sequential Batch Leachate Testing

<u>Cycle</u>	<u>Anaerobic</u>		<u>Aerobic</u>	
	<u>pH</u>	<u>Conductivity</u>	<u>pH</u>	<u>Conductivity</u>
1	5.6(0.12)*	0.08(0.003)	3.9(0)	0.51(0.06)
2	5.3(0.03)	0.10(0.01)	3.9(0.03)	0.19(0)
3	5.7(0.06)	0.15(0.01)	4.0(0.03)	0.11(0)
4	5.6(0.09)	0.16(0)	3.9(0.03)	0.07(0.09)
5	5.5(0.03)	0.12(0)	4.0(0.09)	0.04(0)
6	5.6(0.03)	0.12(0.01)	4.0(0.06)	0.05(0.01)
7	5.7(0.03)	0.10(0.01)	4.3(0.07)	0.06(0.03)

* Standard error in parentheses (N = 3).

lower (3.9 to 4.3) than anaerobic leachate pH (5.3 to 5.7), but was stable over seven leach cycles. Conductivity of oxidized sediment leachate was higher than that of anaerobic sediments. Leachate pH and conductivity changes over seven leach cycles under anaerobic and aerobic conditions are illustrated in Figure 3.

Anaerobic metal releases

22. Linear partitioning isotherms between steady-state sediment and leachate concentrations were not significant for any of the metals, PAHs, or TRPHs tested in anaerobic Hamlet City Lake sediment. Therefore, distribution coefficients describing the partitioning of metals, PAH compounds, and TRPH between sediment and water over the entire sequence of leaching cycles could not be computed. Steady-state metal concentrations in leachate and sediment, obtained from the sequential batch leaching tests for anaerobic sediment, are presented in Tables 8 and 9, respectively. Arsenic and zinc exhibited peak leachate concentrations in the first leach cycle, then decreased. Copper and lead concentrations peaked in the third leach cycle, while nickel concentrations peaked during the second leach cycle. Cadmium concentrations remained relatively constant during the leachate test, dropping below 0.0006 mg/L only during the fourth and seventh leach cycles. Mercury was below detection limits (<0.0004 mg/L) in all leachate samples.

Aerobic metal releases

23. Steady-state metal concentrations in leachate and sediment obtained from the sequential batch leaching tests for aerobic sediment are presented in Tables 10 and 11, respectively. Significant relationships existed between steady-state sediment and leachate concentrations for Zn, Cu, Ni, Cd, and Pb in Hamlet City Lake sediment. Distribution coefficients, K_d , describing the partitioning of metals under aerobic conditions between sediment and water and regression coefficients determined using the Statistical Analysis System are presented in Table 12. Distribution coefficients ranged from 3.29 for Cu to 8.47 for Zn, values similar to those observed during anaerobic leaching of Indiana Harbor sediment, another freshwater sediment. As was the case for anaerobic sediment, Hg was below detection limits (<0.0002 mg/L) in the aerobic leachate.

24. Leachate metal concentrations during sequential batch leaching of anaerobic and aerobic Hamlet City Lake sediment are presented in Figure 4 for Zn, Cu, Pb, Ni, Cd, and As. Despite the drop in pH as a result of sediment

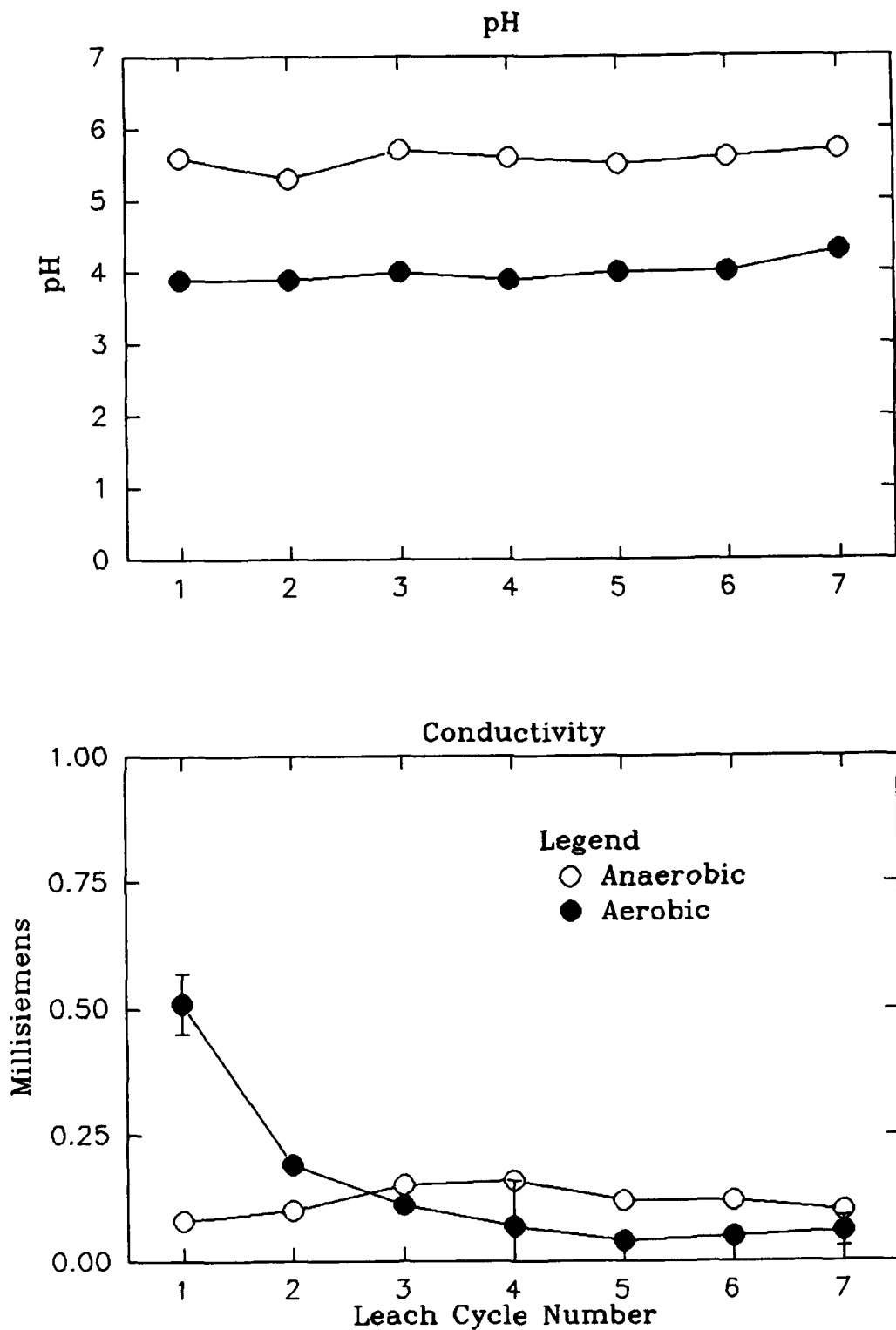


Figure 3. Sequential batch leachate pH and conductivity in anaerobic and aerobic Hamlet City Lake sediment

Table 8
Leachate Heavy Metal Concentrations* in Anaerobic Hamlet City Lake
Sediment for Seven Leach Cycles

Parameter	Leach Cycle						
	1	2	3	4	5	6	7
As	0.067(0.005)	0.052(0.01)	0.034(0.001)	0.009(0.005)	<0.010	<0.010	0.037(0.006)
Zn	0.451(0.38)	0.150(0.031)	0.151(0.031)	0.076(0.001)	0.104(0.016)	0.091(0.009)	0.050(0.017)
Cu	0.028(0.005)	0.069(0.011)	0.112(0.015)	0.003(0.003)	0.004(0.002)	0.021(0.017)	0.046(0.021)
Ni	0.016(0.01)	0.021(0.01)	0.009(0.003)	<0.010	0.02(0.009)	0.01(0.004)	<0.010
Pb	0.034(0.004)	0.20(0.05)	0.32(0.05)	0.02(0.003)	0.02(0.001)	0.03(0.01)	0.10(0.08)
Cd	0.0009(0.0003)	0.005(0.002)	0.0006(0.0001)	0.0002(0)	0.0006(0.0002)	0.0007(0.0001)	0.0003(0)

* Expressed in milligrams per liter (standard error in parentheses) (N = 3).

Table 9
Sediment Heavy Metal Concentrations* in Anaerobic
Hamlet City Lake for Seven Leach Cycles

Parameter	Leach Cycle						
	1	2	3	4	5	6	7
As	12.1(0.02)	11.9(0.02)	11.8(0.01)	11.8(0.01)	11.8(0.01)	11.8(0.01)	11.6(0.03)
Zn	140.2(1.5)	139.6(1.40)	139.0(1.32)	138.7(1.31)	138.3(1.37)	137.9(1.40)	137.7(1.36)
Cu	201.9(0.02)	201.6(0.03)	201.2(0.08)	201.2(0.09)	201.1(0.09)	201.1(0.15)	200.9(0.23)
Ni	12.4(0.03)	12.4(0.02)	12.3(0.01)	12.3(0.01)	12.2(0.03)	12.2(0.02)	12.2(0.02)
Pb	215.9(0.02)	215.1(0.19)	213.8(0.38)	213.7(0.39)	213.6(0.39)	213.5(0.40)	213.1(0.66)
Cd	0.58(0.01)	0.56(0.01)	0.55(0.01)	0.55(0.01)	0.55(0.005)	0.55(0.004)	0.55(0.005)

* Expressed in milligrams per kilogram (standard error in parentheses) (N = 3).

Table 10

Leachate Heavy Metal Concentrations* in Aerobic Hamlet City LakeSediment for Seven Leach Cycles

Parameter	Leach Cycle						
	1	2	3	4	5	6	7
As	<0.0020	<0.0020	<0.0020	<0.0020	<0.0020	<0.0020	<0.0020
Zn	0.939(0.02)	0.517(0.01)	0.351(0.02)	0.284(0.04)	0.229(0.02)	0.199(0.02)	0.200(0.02)
Cu	0.022(0.003)	0.008(0.001)	0.009(0.002)	<0.050	0.005(0.003)	0.002(0.002)	<0.050
Ni	0.072(0.001)	0.026(0.01)	0.019(0.01)	0.02(0.001)	0.02(0)	0.01(0)	0.01(0)
Pb	0.06(0.002)	0.03(0.003)	0.02(0.001)	0.01(0.001)	0.015(0.005)	0.009(0.001)	0.007(0.00004)
Cd	0.009(0)	0.005(0)	0.003(0)	0.002(0)	0.0020	0.002(0)	0.001(0)

* Expressed in milligrams per liter (standard error in parentheses)(N = 3).

Table 11

Sediment Heavy Metal Concentrations* in AerobicHamlet City Lake for Seven Leach Cycles

Parameter	Leach Cycle						
	1	2	3	4	5	6	7
As	12.4(0)	12.4(0)	12.4(0)	12.4(0)	12.4(0)	12.4(0)	12.4(0)
Zn	138.2(0.09)	136.2(0.04)	134.8(0.14)	133.6(0.28)	132.7(0.37)	131.9(0.44)	131.1(0.47)
Cu	201.9(0.01)	201.9(0.01)	201.8(0.02)	201.8(0.02)	201.8(0.02)	201.8(0.01)	201.8(0.01)
Ni	12.21(1.65)	12.10(1.62)	12.03(1.61)	11.95(1.61)	11.87(1.61)	11.82(1.61)	11.77(1.61)
Pb	215.8(0.01)	215.7(0.02)	215.6(0.02)	215.6(0.02)	215.5(0.02)	215.5(0.03)	215.4(0.03)
Cd	0.543(0.001)	0.524(0.001)	0.511(0.001)	0.502(0.002)	0.495(0.002)	0.489(0.003)	0.483(0.003)

* Expressed in milligrams per kilogram (standard error in parentheses)(N = 3).

Table 12
Distribution Coefficients (K_d) and Regression Coefficients
(r^2) for Aerobic Hamlet City Lake Sediment

<u>Metal</u>	<u>K_d</u>	<u>r^2</u>
Zn	8.47	0.803*
Cu	3.29	0.420*
Ni	4.02	0.203*
Cd	6.92	0.866*
Pb	5.97	0.700*

* $P < 0.05$ (95-percent confidence level).

oxidation, leachate concentrations were higher in aerobic compared to anaerobic leachate only for Zn, Ni, and Cd. The As, Cu, and Pb concentrations were higher in anaerobic leachate.

Anaerobic PAH releases

25. Steady-state PAH concentrations in leachate and sediment obtained from the sequential batch leaching tests for anaerobic sediment are presented in Tables 13 and 14, respectively. PAH compounds, when detected in the leachate, were generally present in trace amounts. Of the PAH compounds present in the sediment, only phenanthrene and pyrene were detected in the leachate.

Aerobic PAH releases

26. Only pyrene in trace amounts was detected under aerobic conditions in the leachate. Steady-state pyrene concentrations in leachate and sediment obtained from the sequential batch leaching tests for aerobic sediment are presented in Tables 15 and 16, respectively.

Anaerobic and aerobic TRPH releases

27. Steady-state TRPH concentrations in leachate and sediment obtained from the sequential batch leaching tests for anaerobic and aerobic Hamlet City Lake sediment are presented in Table 17. Concentrations of TRPH in the anaerobic leachate peaked during the second leach cycle. Concentrations of TRPH were below detection limits in the aerobic leachate. Distribution coefficients could not be obtained for TRPH in anaerobic leachate because of the lack of statistical relationships between steady-state sediment and leachate concentrations.

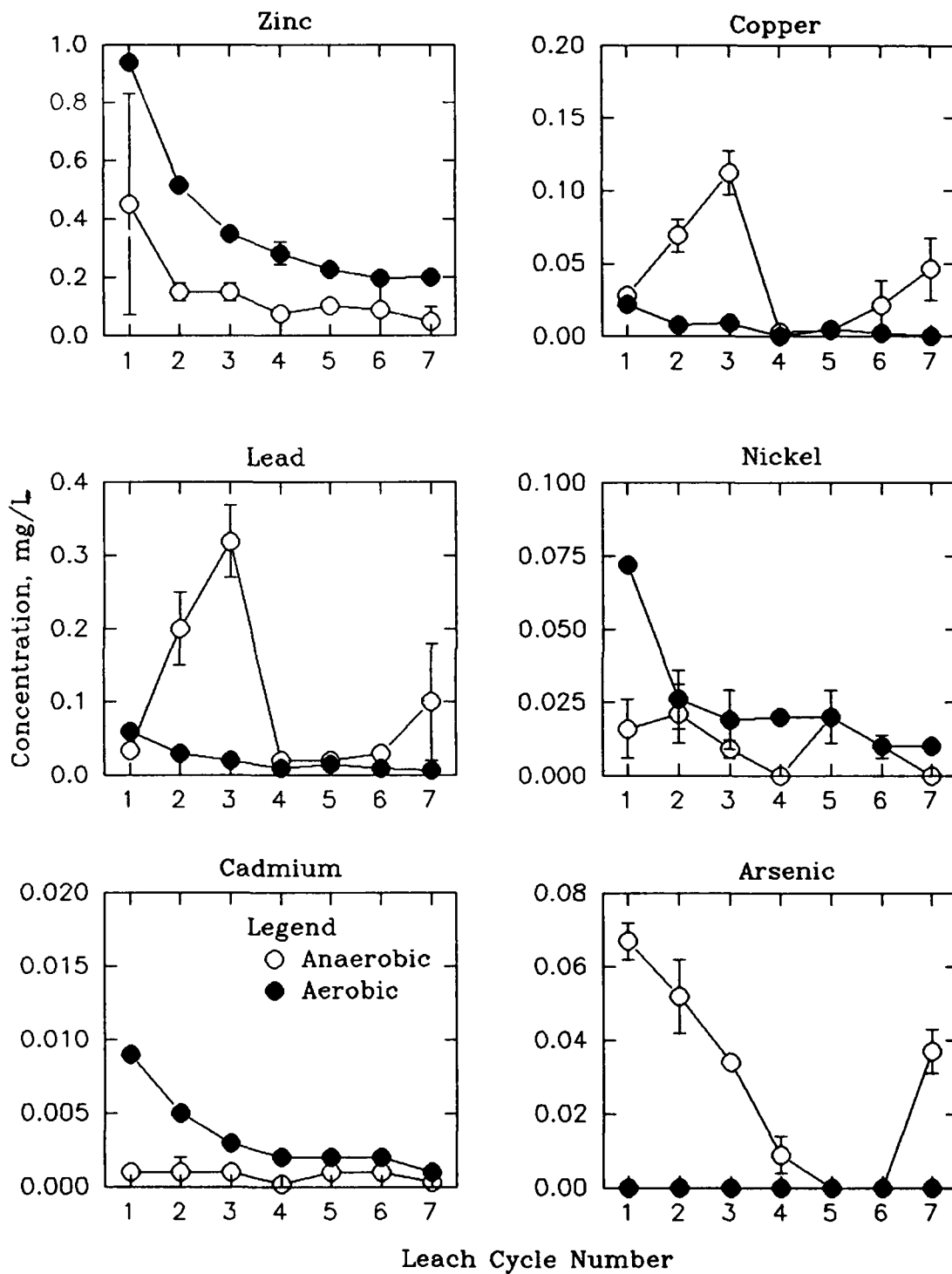


Figure 4. Leachate metal concentrations under anaerobic and aerobic conditions

Table 13

PAH Concentrations* in Leachate from Anaerobic Hamlet City Lake
Sediment for Seven Leach Cycles

<u>Parameter</u>	<u>Leach Cycle</u>						
	<u>1</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>	<u>6</u>	<u>7</u>
Phenanthrene	<0.00064	0.0001 (0)J**	<0.00087	<0.00095	<0.00078	<0.00078	<0.00078
Pyrene	0.0002 (0.0001)	0.0004 (0.0001)	0.0001 (0.0001)	<0.00040	0.0001 (0)	0.0001 (0)	0.0001 (0)

* Expressed in milligrams per liter (standard error in parentheses) (N = 3).

** Value below Method Detection Limit (present in trace amounts).

Table 14

PAH Sediment Concentrations* in Anaerobic Hamlet City Lake
for Seven Leach Cycles

<u>Parameter</u>	<u>Leach Cycle</u>						
	<u>1</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>	<u>6</u>	<u>7</u>
Phenanthrene	1.40 (0.0002)	1.40 (0)	1.399 (0.0002)	1.399 (0.0002)	1.399 (0.0002)	1.399 (0.0002)	1.399 (0.0002)
Pyrene	3.129 (0.0004)	3.128 (0.001)	3.128 (0.0011)	3.127 (0.001)	3.127 (0.001)	3.127 (0.001)	3.126 (0.001)

* Expressed in milligrams per kilogram (standard error in parentheses) (N = 3).

Table 15

Pyrene Concentrations* in Leachate from Aerobic Hamlet City Lake
Sediment for Seven Leach Cycles

<u>Parameter</u>	<u>Leach Cycle</u>						
	<u>1</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>	<u>6</u>	<u>7</u>
Pyrene	0.0002J** (0.001)	0.0004J (0.001)	0.0001J (0.0001)	<0.00016J	0.0001J (0)	0.0001J (0)	0.0001J (0)

* Expressed in milligrams per liter (standard error in parentheses)(N = 3).

** Value below Method Detection Limits (present in trace amounts).

Table 16

Pyrene Sediment Concentrations* in Aerobic Hamlet City Lake
for Seven Leach Cycles

<u>Parameter</u>	<u>Leach Cycle</u>						
	<u>1</u>	<u>2</u>	<u>3</u>	<u>4</u>	<u>5</u>	<u>6</u>	<u>7</u>
Pyrene	1.969 (0.0004)	1.968 (0.0009)	1.968 (0.001)	1.967 (0.001)	1.967 (0.0009)	1.967 (0.0008)	1.966 (0.0007)

* Expressed in milligrams per kilogram (standard error in parentheses)(N = 3).

Table 17

TRPH Concentrations in Leachate* and Sediment** from Anaerobic and
Aerobic Hamlet City Lake Sediment for Seven Leach Cycles

<u>Leach Cycle</u>	<u>Anaerobic</u>		<u>Aerobic</u>	
	<u>Leachate</u>	<u>Sediment</u>	<u>Leachate</u>	<u>Sediment</u>
1	1.63(0.26)	13,993.5(1.04)	<0.8	2,200
2	7.87(1.43)	13,962.0(9.43)	<0.6	2,200
3	3.17(1.43)	13,949.3(4.44)	<0.6	2,200
4	1.00(1.00)	13,945.3(6.41)	<0.5	2,200
5	<0.7	13,945.3(6.41)	<0.5	2,200
6	0.33(0.33)	13,944.0(6.84)	<0.5	2,200
7	<0.8	13,944.0(6.84)	<0.5	2,200

* Expressed in milligrams per liter (standard error in parentheses) (N = 3).

** Expressed in milligrams per kilogram (standard error in parentheses).

Cumulative and Percentage Losses of Metals,
PAHs, and TRPHs During Leaching

28. Cumulative net mass release of metals from Hamlet City Lake sediment was approximately 2.5 times higher for Zn, Ni, and Cd under aerobic than under anaerobic conditions (Table 18). Net mass releases of As, Cu, and Pb were higher under anaerobic conditions. Mercury was not released under either anaerobic or aerobic conditions. Releases of TRPH, phenanthrene, and pyrene were higher under anaerobic conditions than under aerobic conditions. Total concentrations of these constituents in sediment were lower in oxidized sediment than in anaerobic sediment. Percents of metals, PAHs, and TRPHs released into the leachate ranged from 0.00 for phenanthrene, pyrene, and TRPH under aerobic conditions to a high of 16.7 percent for Cd loss under aerobic conditions.

Column Leaching Results

Column operating parameters

29. Column operating parameters for metals and organics showed no substantial differences (Table 19). As previously discussed, the use of separate

Table 18

Cumulative Mass Loss ($\mu\text{g/g}$) and Percent Loss of Metals, PAHs, and
TRPHs from Hamlet City Lake Sediments Following
Seven Leach Cycles

<u>Parameter</u>	<u>Cumulative Mass Loss, $\mu\text{g/g}$</u>		<u>Percent Loss of Initial Concentration</u>	
	<u>Anaerobic</u>	<u>Aerobic</u>	<u>Anaerobic</u>	<u>Aerobic</u>
As	0.80	0.00	6.43	0.00
Zn	4.29	10.9	3.02	7.68
Cu	1.13	0.20	0.56	0.10
Ni	0.29	0.73	2.28	5.84
Cd	0.035	0.097	5.98	16.72
Hg	0.00	0.00	0.00	0.00
Pb	2.92	0.60	1.35	0.28
TRPH	56.0	0.00	0.40	0.00
Phenanthrene	0.001	0.00	0.07	0.00
Pyrene	0.004	0.004	0.13	0.20

Table 19

Column* Operating Parameters

<u>Column</u>	<u>Parameter</u>	<u>n**</u>	<u>SG†</u>	<u>w††</u>	<u>v‡</u>	<u>D_p‡‡</u>
1	Metals	0.81	2.33	1.890	7.0E-06	1.0E-04
2	Metals	0.81	2.33	1.890	6.3E-06	9.0E-05
4	Metals	0.81	2.33	1.890	4.4E-06	6.3E-05
13	PAHs	0.81	2.33	1.890	9.3E-06	1.3E-04
14	PAHs	0.81	2.33	1.890	9.0E-06	1.3E-04
15	PAHs	0.81	2.33	1.890	1.1E-05	1.6E-04

* Length, 4 cm.

** Porosity.

† Specific gravity.

†† Water content, weight of water/weight of solids.

‡ Average pore water velocity, cm/sec.

‡‡ Dispersion coefficient, cm^2/sec .

columns for metals and organics is primarily one of convenience for collection and preservation of samples.

30. Column length is fixed by the column geometry and is therefore the same for all columns. Porosity, specific gravity, and water content were determined on one sample collected during loading of the columns using methods given in U.S. Army Corps of Engineers (1970). The dispersion coefficient was determined by running a salt tracer study on one column after all the leachate samples for chemical analysis were collected. The tracer study is described in Appendix B. Average pore water velocities (obtained from column operating records) were about the same for all the columns.

Data presentation format

31. Column leachate data are reported as a function of the number of pore volumes eluted from the columns. One pore volume is that volume in the sediment chamber (Figure 2) occupied by water. Since the columns were operated in a saturated condition, all the voids were filled by water. Figure 5 illustrates the pore volume concept. For saturated conditions, the pore volume is given by

$$P_v = ALn \quad (1)$$

where

P_v = pore volume of the sediment, cm^3

A = cross-sectional surface area of the sediment column, cm^2

L = length of the sediment column, cm

n = porosity, dimensionless

32. The number of pore volumes eluted is the cumulative volume of water collected divided by P_v . For the steady-flow conditions maintained during column leaching, the number of pore volumes eluted is also given by

$$T = \frac{Qt}{P_v} = \frac{AVnt}{ALn} = \frac{V_{dt}}{Ln} \quad (2)$$

where

T = pore volumes eluted

Q = flow, cm^3/sec

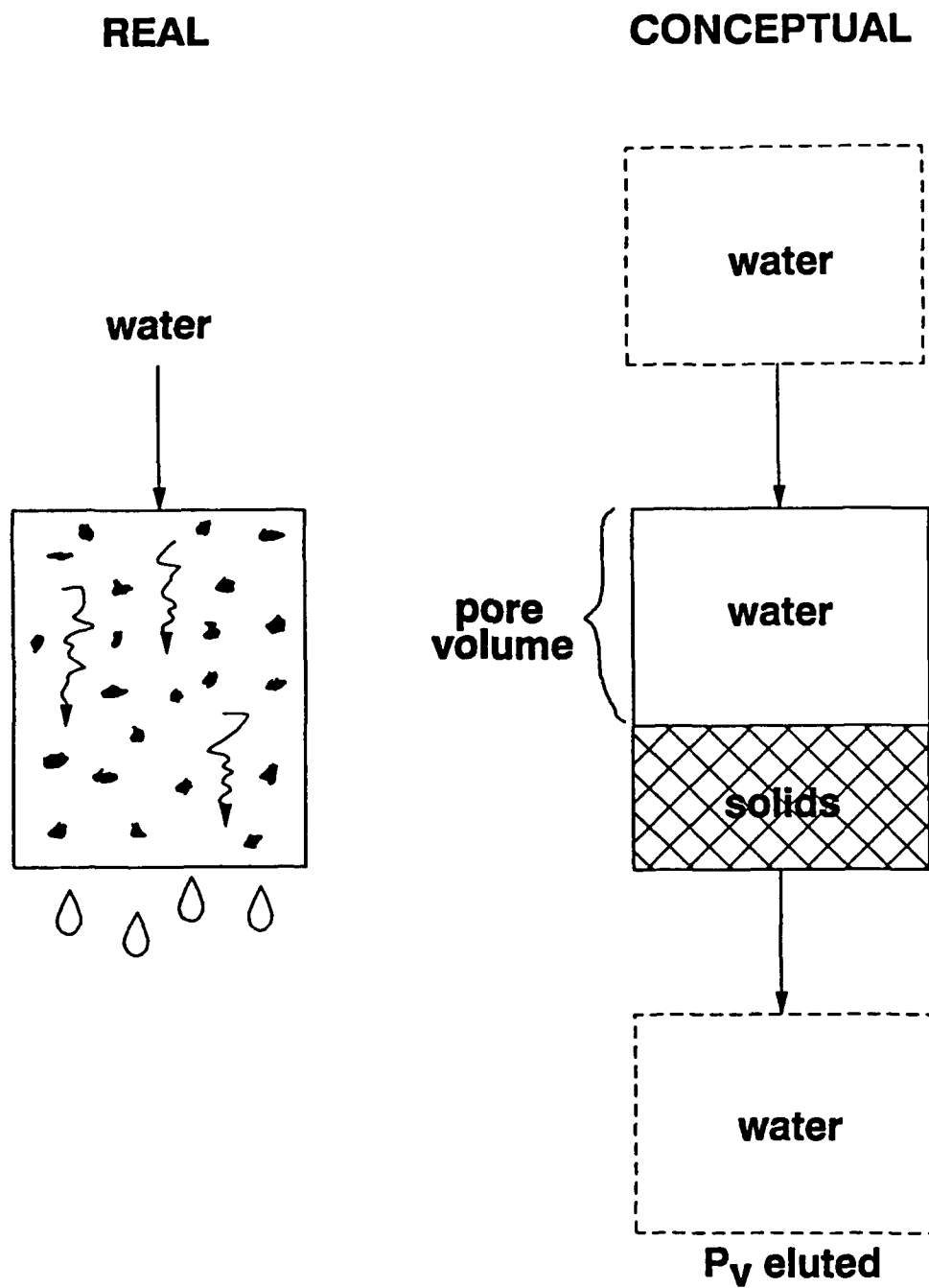


Figure 5. Pore volume (P_v) definition sketch

t = time, sec

V = average pore water velocity, cm/sec

V_d = Darcy velocity, cm/sec

33. The field time for elution of one pore volume is given by

$$t' = \frac{Ln}{V_d} \quad (3)$$

where t' is the time to elute one pore volume ($T = 1$). For saturated dredged material with no standing water and vertically downward flow, the Darcy velocity is equal to the hydraulic conductivity of the dredged material. For saturated dredged material in an upland disposal facility of area A , the field elution time for one pore volume of water is given by

$$t' = \frac{ALn}{AK} = \frac{Ln}{K} \quad (4)$$

where

L = depth of dredged material fill, cm

K = hydraulic conductivity, cm/sec

For example, if $L = 457$ cm, $K = 1 \text{ E-}06$ cm/sec, and $n = 0.6$, the field elution time for one pore volume is about 9 years.

34. The discussion above illustrates how to convert laboratory time for column leaching studies to an equivalent field time when the hydraulic gradient is equal to 1. In the field, hydraulic gradients are usually substantially less than 1. For this reason, Equation 4 provides an estimate of the minimum amount of time required to elute one pore volume. Because the sediment pore volume is the basis for relating laboratory column time to field time, it is therefore convenient to present column elution curves as contaminant concentration versus pore volumes eluted.

General column leachate quality

35. The pH of leachate from columns operated for testing of metals (Figure 6) and organics (Figure 7) showed little variation and remained near pH 6.0. Column leachate pH was about the same as the pH of anaerobic batch

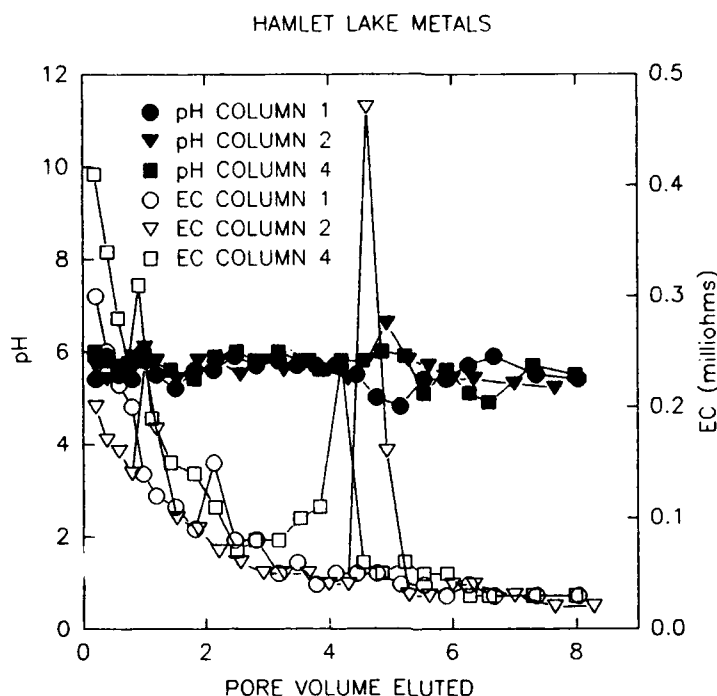


Figure 6. Electrical conductivity and pH in column leachates collected for metals analysis

leachate (Table 7). In both cases, pH was generally steady throughout the leaching procedure.

36. The electrical conductivity (EC) of column leachates from columns operated for metals analysis showed classical washout trends as expected. Since EC is a measure of dissolved ions, a washout curve is expected. The EC versus pore volumes eluted plot should monotonically decrease; that is, electrical conductivity should never increase. The increases in electrical conductivity between three and six pore volumes eluted in columns 2 and 4 suggest that isolated pockets in the sediment through which water had not been flowing opened up and began to transmit water. The EC curves for the columns operated for PAH analysis (Figure 7) do not show the initial washout of dissolved ions that is shown in Figure 6 because samples for electrical conductivity were not taken until after two pore volumes were eluted. The electrical conductivity curves from columns conducted for PAH analysis also show evidence of initially isolated pockets that begin to transmit water well after several pore volumes are eluted.

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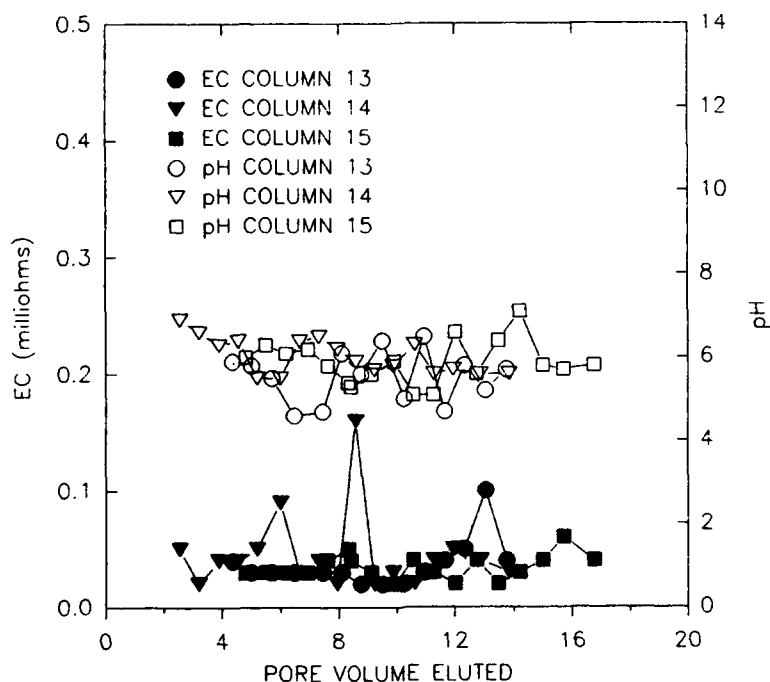


Figure 1. Electrical conductivity and pH in column leachates collected for PAH analysis

37. Total organic carbon elution curves from columns operated for metals analysis are shown in Figure 8. TOC was not measured in leachates from columns operated for PAH analysis because samples were collected under a hexane layer. TOC elution from column 1 was initially highly variable and then became steady at very low concentrations (approximately 10 mg/L). TOC elution from columns 2 and 4 was initially low (approximately 12 mg/L) and remained low through the column leach test.

Metals in column leachates

38. The highest metals concentrations generally occurred before one pore volume was eluted and thereafter decreased to concentrations below detectable limits (Tables 20-22). Depletion of initial concentrations during column leaching suggests that the sediment reservoir of leachable metals is small and initial metals concentrations in dredged material pore water will be reduced by convective transport to levels that are difficult to measure. Lead and mercury were exceptions to this general trend. Column elution trends for each metal are described below.

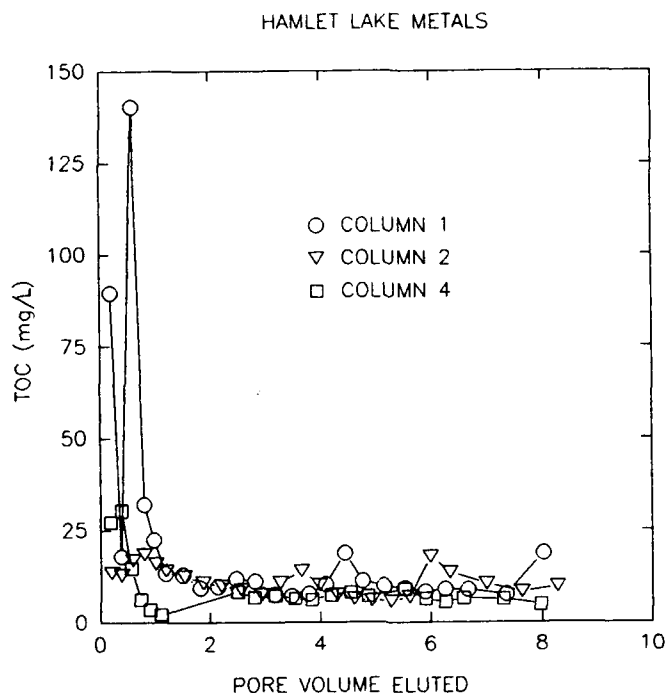


Figure 8. Total organic carbon in leachate from columns conducted for metals analysis

39. Arsenic concentrations dropped from initial values around $0.01 \mu\text{g/L}$ to concentrations that were consistently less than $0.005 \mu\text{g/L}$. This type of elution curve suggests that initial arsenic concentrations in dredged material pore water will be reduced by convective transport and the reservoir of leachable arsenic in the sediment solids is insufficient to maintain concentrations above 0.005 mg/L .

40. Cadmium concentrations dropped from initial values between 0.6 and $2.8 \mu\text{g/L}$ to concentrations that were consistently less than $0.1 \mu\text{g/L}$. This type of elution curve suggests that initial cadmium concentrations in dredged material pore water will be reduced by convective transport and the reservoir of leachable cadmium in the sediment solids is insufficient to maintain concentrations above $0.1 \mu\text{g/L}$ at constant low-flow conditions.

41. Copper concentrations dropped from initial values between 0.008 and 0.164 mg/L to concentrations that were consistently less than $0.01 \mu\text{g/L}$. This type of elution curve suggests that initial copper concentrations in dredged material pore water will be reduced by convective transport and the reservoir

Table 20
Metals Concentrations (mg/L) in Leachate from Column 1

Pore Volume Eluted	Arsenic	Cadmium	Copper	Lead	Mercury	Nickel	Zinc
0.10	0.0119	0.0028	0.164	0.0287	<0.0004	0.108	0.465
0.30	0.0039	0.0001	0.006	0.0098	<0.0004	0.049	0.019
0.49	0.0024	0.0003	0.004	0.0076	<0.0004	0.031	<0.008
0.69	<0.002	<0.0001	0.006	0.0056	<0.0004	0.011	<0.008
0.89	<0.002	<0.0001	0.001	0.0049	<0.0004	<0.001	<0.008
1.08	<0.002	<0.0001	0.001	0.0079	<0.0004	<0.001	<0.008
1.35	<0.002	0.0001	0.001	0.0052	<0.0004	<0.001	<0.008
1.66	<0.002	0.0001	0.001	0.0068	<0.0004	<0.001	<0.008
1.97	<0.002	<0.0001	0.001	0.0043	<0.0004	<0.001	<0.008
2.30	<0.002	<0.0001	0.001	0.0050	<0.0004	<0.001	<0.008
2.64	<0.002	<0.0001	0.005	<0.0004	<0.0004	<0.001	<0.008
2.99	<0.002	<0.0001	<0.001	0.0041	<0.0004	<0.001	<0.008
3.32	<0.002	<0.0001	0.001	0.0054	<0.0004	<0.001	<0.008
3.63	<0.002	<0.0001	0.001	0.0059	<0.0004	<0.001	<0.008
3.94	<0.005	<0.0001	0.007	0.0050	<0.0004	0.004	<0.005
4.27	<0.005	<0.0001	0.006	0.0050	<0.0004	0.004	<0.005
4.61	<0.005	<0.0001	0.006	0.0480	<0.0004	0.003	<0.005
4.96	<0.005	<0.0001	0.012	0.0060	<0.0004	0.004	<0.005
5.35	<0.005	<0.0001	0.005	0.0060	<0.0004	0.003	<0.005
5.72	<0.005	<0.0001	0.005	0.0050	<0.0004	0.003	<0.005
6.07	0.006	<0.0001	0.008	0.0070	<0.0004	0.004	<0.005
6.46	<0.005	<0.0001	0.007	0.0070	<0.0004	0.003	<0.005
7.01	<0.005	<0.0001	<0.001	0.0090	<0.0004	0.004	<0.005
7.68	<0.005	<0.0001	0.007	0.0070	<0.0004	0.003	<0.005

Table 21
Metals Concentrations (mg/L) in Leachate from Column 2

Pore Volume Eluted	Arsenic	Cadmium	Copper	Lead	Mercury	Nickel	Zinc
0.10	0.0048	0.0007	0.079	0.0161	<0.0004	0.038	0.048
0.29	0.0021	0.0001	0.022	0.0043	<0.0004	0.019	0.021
0.49	0.0072	0.0001	0.015	0.0179	<0.0004	0.036	0.046
0.70	0.0067	0.0002	0.006	0.0142	<0.0004	0.027	0.022
0.91	0.0043	<0.0001	0.006	0.0106	<0.0004	0.014	0.008
1.10	0.0036	<0.0001	0.001	0.0084	<0.0004	0.009	<0.008
1.36	0.0028	0.0001	<0.001	0.0051	<0.0004	0.009	<0.008
1.70	0.0028	0.0001	0.001	0.0068	<0.0004	0.005	<0.008
2.04	<0.0020	<0.0001	0.002	0.0090	<0.0004	0.004	<0.008
2.38	<0.0020	<0.0001	0.002	0.0107	<0.0004	0.002	<0.008
2.74	<0.0020	<0.0001	0.002	0.0103	<0.0004	<0.001	<0.008
3.10	<0.0020	<0.0001	0.004	0.0112	<0.0004	<0.001	<0.008
6.46	0.0050	<0.0001	0.006	0.0090	<0.0004	0.003	<0.005
3.82	<0.0050	<0.0001	0.006	0.0100	<0.0004	0.002	<0.005
4.14	<0.0050	<0.0001	0.008	0.0080	<0.0004	0.002	<0.005
4.46	<0.0050	<0.0001	0.006	0.0080	<0.0004	0.005	<0.005
4.77	<0.0050	<0.0001	0.006	0.0090	<0.0004	0.003	<0.005
5.11	<0.0050	<0.0001	0.006	0.0090	<0.0004	<0.001	<0.005
5.45	<0.0050	<0.0001	<0.001	0.0100	<0.0004	<0.001	<0.005
5.80	<0.0050	0.0001	<0.001	0.0110	<0.0004	<0.001	<0.005
6.17	<0.0050	<0.0001	<0.001	0.0090	<0.0004	<0.001	<0.005
6.67	<0.0050	0.0001	<0.001	0.0060	<0.0004	<0.001	<0.005
7.32	<0.0050	<0.0001	<0.001	0.0060	<0.0004	<0.001	<0.005
7.96	<0.0050	<0.0001	<0.001	0.0040	<0.0004	<0.001	<0.005

Table 22

Metals Concentrations (mg/L) in Leachate from Column 4

<u>Pore Volume Eluted</u>	<u>Arsenic</u>	<u>Cadmium</u>	<u>Copper</u>	<u>Lead</u>	<u>Mercury</u>	<u>Nickel</u>	<u>Zinc</u>
0.09	0.0049	0.0006	0.008	0.0058	<0.0004	0.291	1.050
0.29	0.0068	0.0002	0.010	0.0103	<0.0004	0.098	0.136
0.48	0.0059	<0.0001	0.006	0.0073	<0.0004	0.044	0.010
0.65	0.0031	<0.0001	0.003	0.0063	<0.0004	0.031	<0.008
0.83	0.0020	<0.0001	0.004	0.0071	<0.0004	0.024	<0.008
1.01	<0.0020	0.0001	0.007	0.0074	<0.0004	0.024	<0.008
1.27	<0.0020	<0.0001	0.007	0.0071	<0.0004	0.015	<0.008
1.61	<0.0020	<0.0001	0.008	0.0064	<0.0004	0.012	<0.008
1.98	<0.0020	<0.0001	0.009	0.0066	<0.0004	0.006	<0.008
2.32	<0.0020	<0.0001	0.014	0.0056	<0.0004	0.003	<0.008
2.65	<0.0020	<0.0001	0.007	0.0068	<0.0004	0.005	<0.008
2.99	<0.0020	<0.0001	0.008	0.0073	<0.0004	0.003	<0.008
3.35	<0.0020	0.0004	0.006	0.0068	<0.0004	0.004	<0.008
3.69	<0.0020	<0.0001	0.005	0.0073	<0.0004	0.003	<0.008
4.02	<0.0020	<0.0001	0.005	0.0073	<0.0004	0.002	<0.008
4.37	<0.0020	0.0001	0.006	0.0082	<0.0004	0.001	<0.008
4.70	<0.0020	<0.0001	0.005	0.0089	<0.0004	<0.001	<0.008
5.04	<0.0020	<0.0001	0.009	0.0092	<0.0004	<0.001	<0.008
5.38	0.0050	<0.0001	<0.001	0.0100	<0.0004	<0.001	<0.005
5.72	<0.0050	0.0002	<0.001	0.0070	<0.0004	<0.001	<0.005
6.09	<0.0050	<0.0001	0.004	0.0080	<0.0004	<0.001	<0.005
6.43	<0.0050	<0.0001	<0.001	0.0070	<0.0004	<0.001	<0.005
6.94	<0.0050	<0.0001	<0.001	0.0100	<0.0004	<0.001	<0.005
7.63	0.0050	<0.0001	<0.001	0.0070	<0.0004	<0.001	<0.005

of leachable copper in the sediment solids is insufficient to maintain concentrations above 0.01 mg/L.

42. Lead showed a variable elution history with no distinct trends. Lead concentrations ranged from 0.004 to 0.0287 mg/L. In general, concentrations tended to be steady between 0.005 and 0.01 mg/L through the column leach tests. The column leach data for lead suggest very low but relatively constant levels of leachable lead.

43. Mercury was below the detection limit (0.0004 mg/L) in all samples. These data suggest that mercury is either present at negligible levels in the sediment or does not leach from Hamlet Lake sediment.

44. Nickel concentrations decreased from initial values between 0.038 and 0.291 mg/L to concentrations near or below the detection limit (0.001 mg/L). The column data suggest that initial nickel concentrations in dredged material pore water will be reduced by convective transport and that the reservoir of leachable nickel in the sediment solids is very low and insufficient to maintain concentrations above about 0.005 mg/L.

45. Zinc concentrations rapidly decreased from initial values between 0.048 and 1.05 mg/L to concentrations that were consistently below 0.008 mg/L. These data suggest that initial zinc concentrations in dredged material pore water will be rapidly reduced by convective transport and that the leachable reservoir of zinc is very low.

PAHs in column leachates

46. PAHs were detected only eight times in the 944 PAH analyses conducted on column leachates (Tables 23-25). Anthracene (maximum concentration, 0.0006 mg/L), fluoranthene (maximum concentration, 0.00042 mg/L), and pyrene (maximum concentration, 0.00031 mg/L) were detected. These data indicate that most PAHs leach from Hamlet Lake sediment at levels below detection limits.

Table 23
PAH Concentrations (mg/L) in Leachate from Column 13

Parameter	Pore Volume Eluted										
	1.04	1.69	2.35	2.97	3.57	4.34	4.99	5.70	6.46	7.45	8.10
Naphthalene	<0.000900	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600
Acenaphthylene	<0.001200	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000
Acenaphthene	<0.000900	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600
Fluorene	<0.000110	<0.000180	<0.000190	<0.000180	<0.000190	<0.000180	<0.000190	<0.000180	<0.000190	<0.000180	<0.000190
Phenanthrene	<0.000330	<0.000560	<0.000570	<0.000560	<0.000570	<0.000560	<0.000570	<0.000560	<0.000570	<0.000560	<0.000570
Anthracene	<0.000340	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580
Fluoranthene	<0.000110	<0.000180	<0.000190	<0.000180	<0.000190	<0.000180	<0.000190	<0.000180	<0.000190	<0.000180	<0.000190
Pyrene	<0.000160	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240
Chrysene	<0.000080	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130
Benz(a)anthracene	<0.000007	<0.000011	<0.000016	<0.000011	<0.000016	<0.000011	<0.000016	<0.000011	<0.000016	<0.000011	<0.000016
Benzo(b)fluoranthene	<0.000009	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016
Benzo(k)fluoranthene	<0.000009	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015
Benzo(a)pyrene	<0.000012	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020
Indeno(1,2,3-c,d)pyrene	<0.000022	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038
Dibenzo(a,h)anthracene	<0.000020	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030
Benzo(g,h,i)perylene	<0.000039	<0.000066	<0.000067	<0.000066	<0.000067	<0.000066	<0.000067	<0.000066	<0.000067	<0.000066	<0.000067

Parameter	Pore Volume Eluted									
	8.77	9.49	10.27	10.93	11.63	12.32	13.03	13.74	14.44	15.15
Naphthalene	0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600
Acenaphthylene	0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000	<0.002000
Acenaphthene	0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600	<0.001600
Fluorene	0.000180	<0.000190	<0.000180	<0.000190	<0.000190	<0.000190	<0.000190	<0.000190	<0.000190	<0.000190
Phenanthrene	0.000560	<0.000570	<0.000560	<0.000570	<0.000570	<0.000570	<0.000570	<0.000570	<0.000570	<0.000570
Anthracene	0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580	<0.000580
Fluoranthene	0.000180	<0.000190	0.000420	<0.000190	<0.000190	<0.000190	<0.000190	<0.000190	<0.000190	<0.000190
Pyrene	0.000140	<0.000240	0.000310	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240	<0.000240
Chrysene	0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130	<0.000130
Benz(a)anthracene	0.000011	<0.000016	<0.000011	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016
Benz(b)fluoranthene	0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016	<0.000016
Benz(k)fluoranthene	0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015	<0.000015
Benz(a)pyrene	0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020	<0.000020
Indeno(1,2,3-c,d)pyrene	0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038	<0.000038
Dibenzo(a,h)anthracene	0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030
Benzo(g,h,i)perylene	0.000066	<0.000067	<0.000066	<0.000067	<0.000067	<0.000067	<0.000067	<0.000067	<0.000067	<0.000067

Note: Detection limits vary because of variations in sample volume.

Table 24

PAH Concentrations (mg/L) in Leachate from Column 14

Parameter	Pore Volume Eluted									
	1.23	1.87	2.53	3.19	3.87	4.54	5.19	5.97	6.64	7.30
Naphthalene	<0.000900	<0.000500	<0.001600	<0.001700	<0.001600	<0.001700	<0.001600	<0.001700	<0.001600	<0.001700
Acenaphthylene	<0.001200	<0.002200	<0.002000	<0.002200	<0.002000	<0.002200	<0.002000	<0.002200	<0.002000	<0.002200
Acenaphthene	<0.000900	<0.001700	<0.001600	<0.001700	<0.001600	<0.001700	<0.001600	<0.001700	<0.001600	<0.001700
Fluorene	<0.000110	<0.000200	<0.000190	<0.000200	<0.000190	<0.000200	<0.000190	<0.000200	<0.000190	<0.000200
Phenanthrene	<0.000330	<0.000610	<0.000570	<0.000610	<0.000570	<0.000610	<0.000570	<0.000610	<0.000570	<0.000610
Anthracene	<0.000340	<0.000620	<0.000580	<0.000620	<0.000580	<0.000620	<0.000580	<0.000620	<0.000580	<0.000620
Fluoranthene	<0.000110	<0.000200	<0.000190	<0.000200	<0.000190	<0.000200	<0.000190	<0.000200	<0.000190	<0.000200
Pyrene	0.000170	<0.000260	<0.000240	<0.000260	<0.000240	<0.000260	<0.000240	<0.000260	<0.000240	<0.000260
Chrysene	<0.000080	<0.000140	<0.000130	<0.000140	<0.000130	<0.000140	<0.000130	<0.000140	<0.000130	<0.000140
Benzo(a)anthracene	<0.000007	<0.000013	<0.000016	<0.000013	<0.000016	<0.000013	<0.000016	<0.000013	<0.000016	<0.000013
Benzo(b)fluoranthene	<0.000009	<0.000017	<0.000016	<0.000017	<0.000016	<0.000017	<0.000016	<0.000017	<0.000016	<0.000017
Benzo(k)fluoranthene	<0.000009	<0.000016	<0.000015	<0.000016	<0.000015	<0.000016	<0.000015	<0.000016	<0.000015	<0.000016
Benzo(a)pyrene	<0.000012	<0.000022	<0.000020	<0.000022	<0.000020	<0.000022	<0.000020	<0.000022	<0.000020	<0.000022
Indeno(1,2,3-c,d)pyrene	<0.000022	<0.000041	<0.000038	<0.000041	<0.000038	<0.000041	<0.000038	<0.000041	<0.000038	<0.000041
Dibenzo(a,h)anthracene	<0.000020	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030	<0.000030
Benzo(g,h,i)perylene	<0.000039	<0.000072	<0.000067	<0.000072	<0.000067	<0.000072	<0.000067	<0.000072	<0.000067	<0.000072

Parameter	Pore Volume Eluted									
	8.58	9.22	9.87	10.61	11.27	11.93	12.85	13.85		
Naphthalene	<0.001700	<0.001600	<0.000300	<0.001600	<0.001500	<0.001600	<0.001500	<0.000400		
Acenaphthylene	<0.002200	<0.002000	<0.001900	<0.002100	<0.001900	<0.002100	<0.001900	<0.002100		
Acenaphthene	<0.001700	<0.001600	<0.001500	<0.001600	<0.001500	<0.001600	<0.001500	<0.001600		
Fluorene	<0.000200	<0.000190	<0.000170	<0.000190	<0.000170	<0.000190	<0.000170	<0.000190		
Phenanthrene	<0.000610	<0.000570	<0.000520	<0.000570	<0.000520	<0.000570	<0.000520	<0.000570		
Anthracene	<0.000620	<0.000580	<0.000540	<0.000590	<0.000540	<0.000590	<0.000540	<0.000590		
Fluoranthene	<0.000200	<0.000190	<0.000170	<0.000190	<0.000170	<0.000190	<0.000170	<0.000190		
Pyrene	<0.000260	<0.000240	<0.000220	<0.000240	<0.000220	<0.000240	<0.000220	<0.000240		
Chrysene	<0.000140	<0.000130	<0.000120	<0.000130	<0.000120	<0.000130	<0.000120	<0.000130		
Benzo(a)anthracene	<0.000013	<0.000016	<0.000011	<0.000012	<0.000011	<0.000012	<0.000011	<0.000012		
Benzo(b)fluoranthene	<0.000017	<0.000016	<0.000015	<0.000016	<0.000015	<0.000016	<0.000015	<0.000016		
Benzo(k)fluoranthene	<0.000016	<0.000015	<0.000014	<0.000015	<0.000014	<0.000015	<0.000014	<0.000015		
Benzo(a)pyrene	<0.000022	<0.000020	<0.000019	<0.000021	<0.000019	<0.000021	<0.000019	<0.000021		
Indeno(1,2,3-c,d)pyrene	<0.000041	<0.000038	<0.000035	<0.000038	<0.000035	<0.000038	<0.000035	<0.000038		
Dibenzo(a,h)anthracene	<0.000030	<0.000030	<0.000020	<0.000030	<0.000020	<0.000030	<0.000020	<0.000030		
Benzo(g,h,i)perylene	<0.000072	<0.000067	<0.000062	<0.000068	<0.000062	<0.000068	<0.000062	<0.000068		

Note: Detection limits vary because of variations in sample volume.

Table 25

PAH Concentrations (mg/L) in Leachate from Column 15

Parameter	Pore Volume Eluted									
	1.39	2.39	3.04	3.78	4.79	5.48	6.17	6.92	7.60	8.32
Naphthalene	<0.000100	<0.002400	<0.001600	<0.002400	<0.001100	<0.002400	<0.001600	<0.002400	<0.001600	<0.002400
Acenaphthylene	<0.001000	<0.003000	<0.002100	<0.003000	<0.001400	<0.003000	<0.002100	<0.003000	<0.002100	<0.003000
Acenaphthene	<0.000800	<0.002400	<0.001600	<0.002400	<0.001100	<0.002400	<0.001600	<0.002400	<0.001600	<0.002400
Fluorene	<0.000100	<0.000280	<0.000190	<0.000280	<0.000230	<0.000280	<0.000190	<0.000280	<0.000190	<0.000280
Phenanthrene	<0.000290	<0.000850	<0.000680	<0.000850	<0.000400	<0.000850	<0.000580	<0.000850	<0.000580	<0.000850
Anthracene	<0.000300	<0.000880	<0.000600	<0.000880	<0.000410	<0.000880	<0.000600	<0.000880	<0.000600	<0.000880
Fluoranthene	<0.000100	<0.000280	<0.000190	<0.000280	<0.000130	<0.000280	<0.000190	<0.000280	<0.000190	<0.000280
Pyrene	<0.000120	<0.000050	<0.000240	<0.000050	<0.000170	<0.000370	<0.000240	<0.000370	<0.000240	<0.000370
Chrysene	<0.000070	<0.000200	<0.000130	<0.000200	<0.000090	<0.000200	<0.000130	<0.000200	<0.000130	<0.000200
Benz(a)anthracene	<0.000006	<0.000017	<0.000012	<0.000017	<0.000008	<0.000017	<0.000012	<0.000017	<0.000012	<0.000017
Benz(b)fluoranthene	<0.000008	<0.000024	<0.000016	<0.000024	<0.000011	<0.000024	<0.000016	<0.000024	<0.000016	<0.000024
Benz(k)fluoranthene	<0.000008	<0.000023	<0.000015	<0.000023	<0.000011	<0.000023	<0.000015	<0.000023	<0.000015	<0.000023
Benz(a)pyrene	<0.000010	<0.000030	<0.000021	<0.000030	<0.000014	<0.000030	<0.000021	<0.000030	<0.000021	<0.000030
Indeno(1,2,3-c,d)pyrene	<0.000019	<0.000057	<0.000039	<0.000057	<0.000027	<0.000057	<0.000039	<0.000057	<0.000039	<0.000057
Dibenzo(a,h)anthracene	<0.000010	<0.000040	<0.000030	<0.000040	<0.000020	<0.000040	<0.000030	<0.000040	<0.000030	<0.000040
Benzo(g,h,i)perylene	<0.000034	<0.000101	<0.000069	<0.000101	<0.000047	<0.000101	<0.000069	<0.000101	<0.000069	<0.000101

Parameter	Pore Volume Eluted									
	9.11	9.90	10.55	11.26	12.00	12.72	13.46	14.20	14.97	15.69
Naphthalene	0.002400	<0.001600	<0.002400	<0.001600	<0.002400	<0.001600	<0.001500	<0.001600	<0.001500	<0.001600
Acenaphthylene	0.003000	<0.002100	<0.003000	<0.002100	<0.003000	<0.002100	<0.001900	<0.002100	<0.001900	<0.002100
Acenaphthene	0.002400	<0.001600	<0.002400	<0.001600	<0.002400	<0.001600	<0.001500	<0.001600	<0.001500	<0.001600
Fluorene	0.000280	<0.000190	<0.000280	<0.000190	<0.000280	<0.000190	<0.000170	<0.000190	<0.000170	<0.000190
Phenanthrene	0.000850	<0.000580	<0.000850	<0.000580	<0.000850	<0.000570	<0.000800	<0.000570	<0.000520	<0.000520
Anthracene	0.000880	0.000600	<0.000880	0.000600	<0.000880	<0.000590	<0.000540	<0.000590	<0.000540	<0.000540
Fluoranthene	0.000280	<0.000190	<0.000280	<0.000190	<0.000280	<0.000190	<0.000170	<0.000190	<0.000170	<0.000190
Pyrene	0.000100	<0.000240	<0.000370	<0.000240	<0.000080	<0.000240	<0.000220	<0.000240	<0.000220	<0.000240
Chrysene	0.000200	<0.000130	<0.000200	<0.000130	<0.000200	<0.000130	<0.000120	<0.000130	<0.000120	<0.000130
Benz(a)anthracene	0.000017	<0.000012	<0.000017	<0.000012	<0.000017	<0.000012	<0.000011	<0.000012	<0.000011	<0.000012
Benz(b)fluoranthene	0.000024	<0.000016	<0.000024	<0.000016	<0.000024	<0.000016	<0.000015	<0.000016	<0.000015	<0.000016
Benz(k)fluoranthene	0.000023	<0.000015	<0.000023	<0.000015	<0.000023	<0.000015	<0.000014	<0.000015	<0.000014	<0.000015
Benz(a)pyrene	0.000030	<0.000021	<0.000030	<0.000021	<0.000030	<0.000021	<0.000019	<0.000021	<0.000019	<0.000021
Indeno(1,2,3-c,d)pyrene	0.000057	<0.000039	<0.000057	<0.000039	<0.000057	<0.000038	<0.000035	<0.000038	<0.000035	<0.000038
Dibenzo(a,h)anthracene	0.000040	<0.000030	<0.000040	<0.000030	<0.000040	<0.000030	<0.000020	<0.000030	<0.000020	<0.000030
Benzo(g,h,i)perylene	0.000101	<0.000069	<0.000101	<0.000069	<0.000101	<0.000068	<0.000062	<0.000068	<0.000062	<0.000068

Note: Detection limits vary because of variations in sample volume.

PART IV: DISCUSSION

Sequential Batch Data

47. Slope type distribution coefficients could not be calculated for anaerobic Hamlet City Lake sediment because sequential batch leaching results did not follow classical desorption theory. The appearance of peaks in the leachate concentration data indicated nonconstant sediment geochemistry and nonconstant distribution coefficients during leaching. If constant distribution coefficients existed, contaminant concentrations would not have exceeded initial values.

48. Sequential batch leaching results for aerobic sediments followed classical desorption theory. This resulted in distribution coefficients that were constant during the entire period of sequential batch leaching. The reasons for nonconstant partitioning during anaerobic leaching, as expected of freshwater sediments, is unclear, but does indicate that sediment geochemistry was not constant during leaching.

49. PAHs were found in relatively low concentrations in anaerobic sediment. Oxidation of the sediment resulted in substantial losses of PAHs as well as TRPHs. PAHs and petroleum hydrocarbons are rapidly lost through degradation and volatilization following disposal of dredged material and exposure to the atmosphere. Previous work has shown that up to 86 percent of sediment PAHs can be lost during sediment oxidation (Environmental Laboratory 1987), and that subsequent concentrations of PAH in leachate are generally below analytical detection limits (Environmental Laboratory 1987, Myers and Brannon 1989).

50. Hamlet City Lake sediment exhibited a pronounced drop in pH following oxidation, a phenomenon that has been observed for other sediments (Environmental Laboratory 1987, Myers and Brannon 1989). The oxidation of Hamlet City Lake sediment and decreased pH resulted in increased batch leaching of Zn, Ni, and Cd compared with anaerobic sediment.

Comparison of Anaerobic Sequential Batch and Column Data

51. The declining trends in column leachate concentration for some metals indicate constant distribution coefficients. The anaerobic sequential batch data showed complicated release characteristics for some metals,

indicating nonconstant distribution coefficients. The differences result from the short-term nature of the column leach tests and the long-term nature of the sequential batch leach tests. In terms of the amount of water that has washed the sediment solids, the first two cycles of the seven-cycle batch leach test are equivalent to the entire column leach tests. Contaminant-specific comparisons of anaerobic sequential batch (Table 8) and column leach data (Tables 20-22) are discussed below. Aerobic sequential batch data are not compared with the column data because oxidation-reduction conditions in the aerobic batch test are completely different from those in the column test.

52. The arsenic data from batch and column leach tests are in good agreement over the first six of the seven cycles in the sequential batch leach test. The declining trend in arsenic concentrations over the first six cycles of the sequential batch leach test was confirmed in the column test. In addition, the range in arsenic concentrations is similar in batch and column leachates. Thus, batch and column data are consistent for most of the data available.

53. The sequential batch leach data for cadmium indicated a tendency for cadmium concentrations to remain constant around 0.001 mg/L. Cadmium concentrations in the column leach test were lower than in the batch test and showed a tendency to decline. Thus, the batch and column data for cadmium were inconsistent in terms of trends and were not in good agreement in terms of concentrations.

54. The sequential batch leach data for copper showed nonconstant partitioning with increasing copper concentrations over the first three cycles. The column data did not confirm this increasing trend. Maximum copper concentrations were similar for the batch and column leach tests, 0.112 and 0.154 mg/L, respectively.

55. Batch and column leaching trends for lead were similar. No distinct trends were observed in either test. Lead concentrations tended to be higher in the batch than in column leachate. In cycles two and three of the batch test, lead concentrations (0.20 and 0.23 mg/L, respectively) were significantly higher than any of the column lead concentrations.

56. Mercury concentrations were below the detection limit (0.0004 mg/L) in both column and batch tests. Both tests show that mercury does not leach from anaerobic Hamlet Lake sediment.

57. The sequential batch leach data for nickel showed a complicated desorption process in which concentrations were high in cycles one and two,

low in cycles three and four, again high in cycles five and six, and again low in cycle seven. The column data showed a declining trend in nickel concentrations as more pore volumes were eluted. In addition, initial nickel concentrations were significantly higher in the column test than in any of the cycles of the sequential batch test.

58. Zinc concentrations in sequential batch and column leach tests showed a tendency to decrease. Thus, the tests were consistent in terms of the type of desorption process. Because the column data showed a rapid decrease in zinc concentrations from the initial values, most of the column zinc concentrations are well below the batch zinc concentrations. However, the initial zinc concentrations in two of the columns were similar to the zinc concentration in the first cycle of the batch test. The highest zinc concentration (1.05 mg/L) occurred in the initial leachate sample collected from column 4.

Integrated Approach

Theory

59. An integrated approach (Figure 9) involving predicted and observed column elution curves can be applied to obtain a better understanding of processes affecting leaching of contaminants. Transport models used to develop predicted elution curves commonly describe contaminant transfer between solid and water phases by assuming equilibrium conditions between the water and solid phases as follows:

$$q = K_d C \quad (5)$$

where

q = contaminant concentration in the solid phase, mg/kg

K_d = equilibrium distribution coefficient, L/kg

C = contaminant concentration in the water phase, mg/L

The equilibrium distribution coefficient K_d is sediment and contaminant specific. In most transport models, the distribution coefficient is assumed constant over time and space, although evidence to the contrary exists for some sediments and leaching conditions (Brannon et al. 1991).

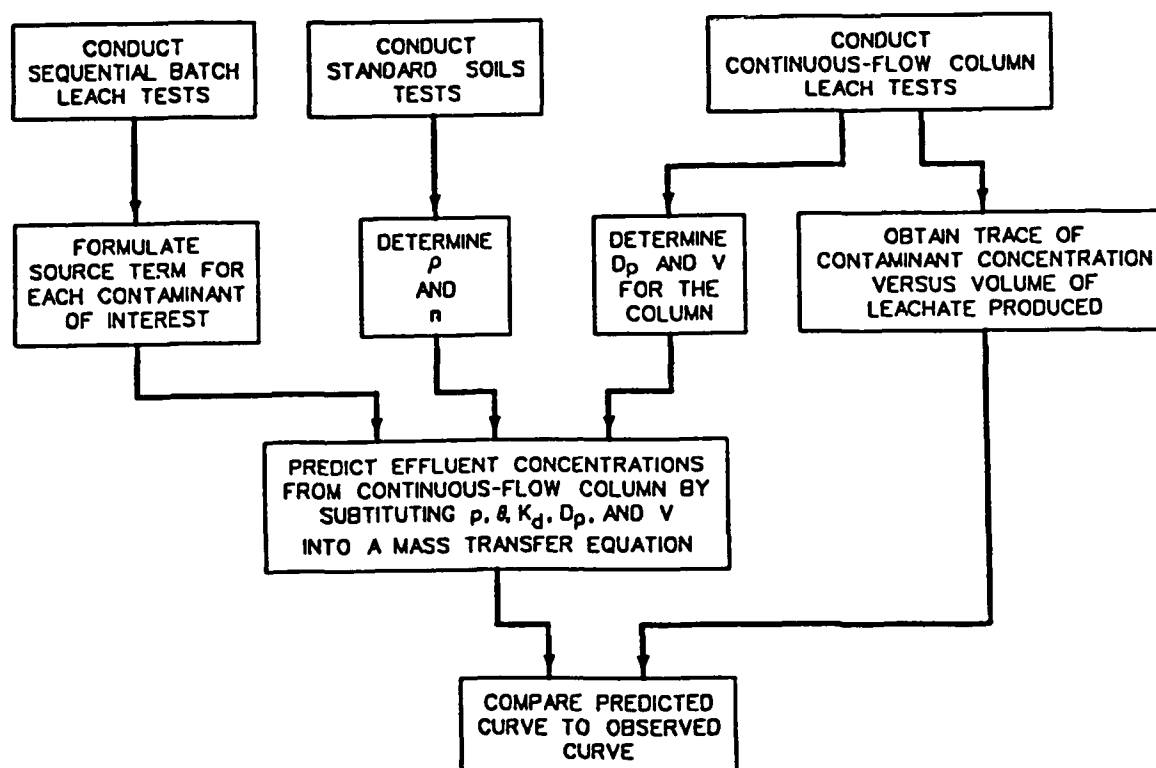


Figure 9. Integrated approach for examining the source term

60. The one-dimensional contaminant transport equation for steady-saturated flow on which the integrated approach is based is given below (Hill, Myers, and Brannon 1988).

$$D_p \frac{\partial^2 C}{\partial z^2} - V \frac{\partial C}{\partial z} + S = \frac{\partial C}{\partial t} \quad (6)$$

$$S = - \frac{\rho_b}{n} \frac{\partial q}{\partial t} \quad (7)$$

where

D_p = dispersion coefficient, cm^2/sec

z = distance, cm

V = average pore water velocity, cm/sec

S = interphase contaminant transfer, $\text{mg}/\text{L}\cdot\text{sec}$

t - time, sec

ρ_b - bulk density, kg/L

n - porosity, cm^3/cm^3

q - sorbed concentration, mg/kg

61. Taking the derivative of Equation 5 with respect to time and assuming K_d is constant yields

$$\frac{\partial q}{\partial t} = \frac{\partial (K_d C)}{\partial t} = K_d \frac{\partial C}{\partial t} \quad (8)$$

62. Equations 6, 7, and 8 can be combined to yield

$$R \frac{\partial C}{\partial t} = D_p \frac{\partial^2 C}{\partial z^2} - V \frac{\partial C}{\partial z} \quad (9)$$

where the retardation factor, R , is defined as follows:

$$R = 1 + \frac{\rho_b K_d}{n} \quad (10)$$

63. For the following initial and boundary conditions

$C(z, 0) = C_0$, C_0 - initial pore water contaminant concentration

$C(0, t) = 0$

$\partial C / \partial t(L, t) = 0$, L - column length

the analytical solution for Equation 9 is (Cleary and Adrian 1973)

$$C(z, t) = \sum_{m=1}^{\infty} \frac{2\beta_m \sin\left(\frac{\beta_m z}{L}\right) \exp\left[\frac{Vz}{2D_p} - \frac{V^2 t}{4D_p R} - \frac{\beta_m^2 D_p t}{L^2 R}\right]}{\beta_m^2 + \frac{VL^2}{2D_p} + \frac{VL}{2D}} \quad (11)$$

where the eigenvalues β_m are the positive roots of the equation

$$\beta_m \cot(\beta_m) + \frac{VL}{2D_p} = 0 \quad (12)$$

64. Sequential batch leach tests provide estimates of distribution coefficients that are independent of column leach data. From the sequential batch leach data, a table of solid phase contaminant concentrations (q) and water phase contaminant concentrations (C) can be developed and plotted (successive batches have differing q and C concentrations). A plot of q versus C yields a desorption isotherm, the slope of which is the distribution coefficient.

65. Sequential batch leaching of freshwater sediments usually yields desorption isotherms such as shown in Figure 10. Desorption isotherms for metals in freshwater sediments commonly do not go through the origin, but intercept the ordinate at some other point (Myers, Brannon, and Price 1992). The intercept indicates the amount of metal in geochemical phases that is resistant to aqueous leaching. The general equation for the type of desorption isotherms shown in Figure 10 is

$$q = K_d C + q_r \quad (13)$$

where q_r is the solid phase contaminant concentration resistant to leaching, in milligrams per kilogram.

Application

66. Figure 11 shows desorption isotherm plots prepared from the anaerobic sequential batch leach data for As, Cd, Cu, Pb, Ni, and Zn. Aerobic desorption isotherms are not presented because the continuously flooded conditions in the column leach tests result in anaerobic leaching conditions. None of the isotherms shown in Figure 11 match the ideal model shown in Figure 10 very well. However, as discussed below, distribution coefficients obtained from these plots for selected metals are consistent with the column leach data.

67. The points in each plot shown in Figure 11 represent q - C pairs from cycles in the sequential batch leach test. The highest point relative to the

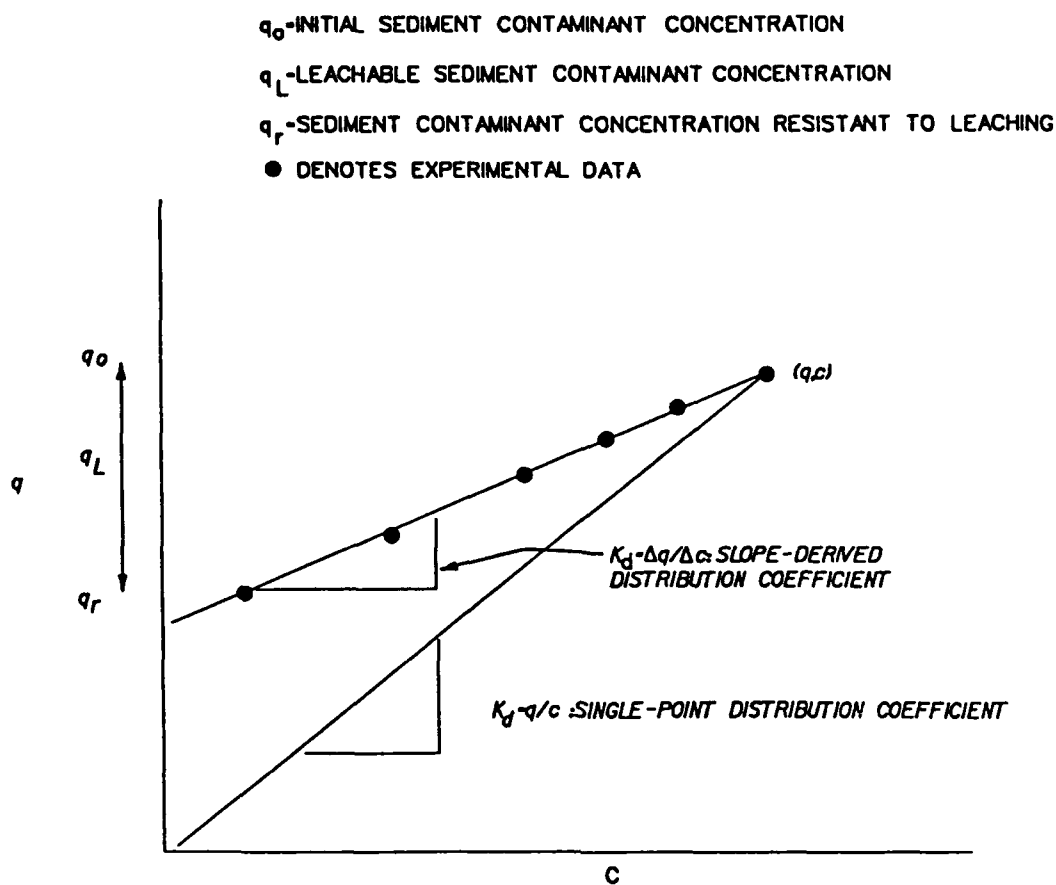


Figure 10. Desorption isotherm illustrating constant partitioning

vertical axis is the data pair from the first cycle, the second highest point relative to the vertical axis is the data pair from the second cycle, and so on. Since each cycle in the sequential batch leach test is equivalent to eluting 4.94 pore volumes of water from a leaching column, the first two cycles in the sequential batch leach test are equivalent to the approximately 10 pore volumes eluted during the column leach test. Complete isotherm analysis is, therefore, not necessary for quantitative comparison of observed elution curves with elution curves predicted using batch-determined distribution coefficients.

68. Distribution coefficients for As, Cd, Ni, and Zn were obtained by first examining desorption isotherms for relative match with the ideal shown in Figure 10 and judging which data pairs to use to calculate distribution coefficients. This type of desorption isotherm analysis involves judgment as to what portions of the isotherms to include in the analysis and is, therefore, highly subjective. Copper and lead isotherms were judged too

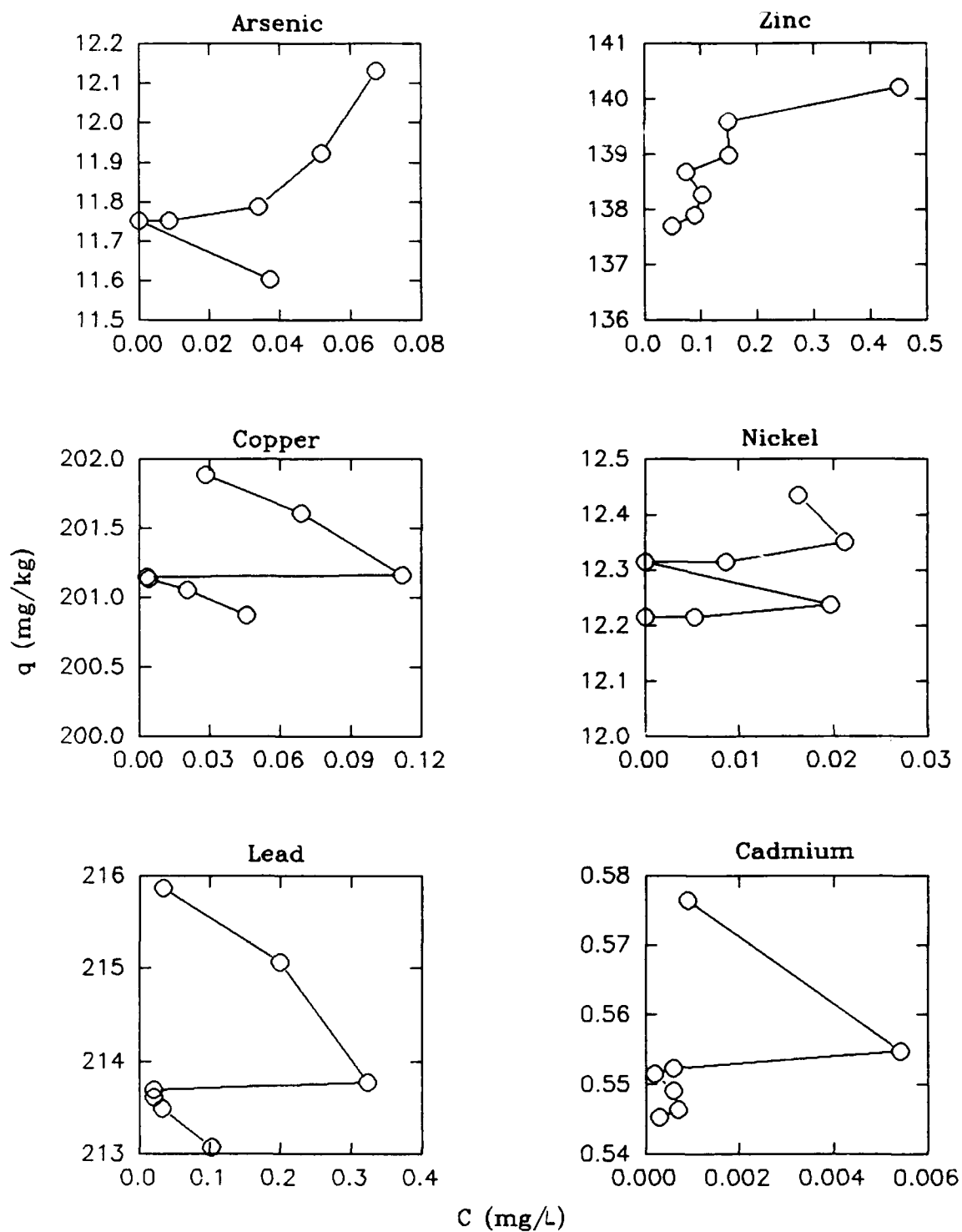


Figure 11. Isotherms for sequential desorption of metals under anaerobic conditions. Plots with less than seven points contain concurrent points

complicated for analysis in terms of the ideal shown in Figure 10. Complicated isotherms showing nonconstant partitioning have been observed and analyzed in leaching studies of estuarine sediments (Lee et al., in preparation). The physical-chemical interpretation applied to estuarine sediments by Lee et al. (in preparation), however, does not have an apparent application for freshwater sediments.

69. Specifics of the desorption isotherm analysis, the distribution coefficients obtained, and comparison of predicted and observed column elution curves for each metal are described in the following sections. Each predicted curve is the result of application of Equation 11 using averaged operating parameters from Table 19 for the three metals columns and metal-specific distribution coefficients.

70. Arsenic. The first six points in the arsenic anaerobic desorption isotherm (Figure 11) show a monotonically decreasing trend. Point six is exactly the same as point five and is not shown. The distribution coefficient calculated using the difference between the first and second points on the desorption isotherm was 13.6 L/kg. Figure 12 shows the arsenic elution curve predicted using this distribution coefficient and observed data.

71. Cadmium. With the exception of the second point, the cadmium anaerobic desorption isotherm (Figure 11) suggests a steep slope, that is, a large distribution coefficient (about 52 L/kg). Large distribution coefficients suggest that leachate concentrations will be low and the low values will persist. The cadmium concentrations in column leachates were low, but initial values did not persist. In addition, distribution coefficients for metals in sediments are usually less than or equal to 10 L/kg (Myers et al., in preparation). Using the second and third points in the cadmium desorption isotherm, the distribution coefficient is 0.5 L/kg. Figure 12 shows the cadmium elution curve predicted using this distribution coefficient and observed data.

72. Nickel. Although the nickel desorption isotherm (Figure 11) is very complicated, the trend over the first four points is similar to the ideal shown in Figure 10. The distribution coefficient calculated using the difference between the first and fourth points was 7.4 L/kg. Figure 12 shows the nickel elution curve predicted using this distribution coefficient and observed data.

73. Zinc. The Langmuir model probably fits the zinc anaerobic isotherm better than the linear model given in Equation 13, but as previously

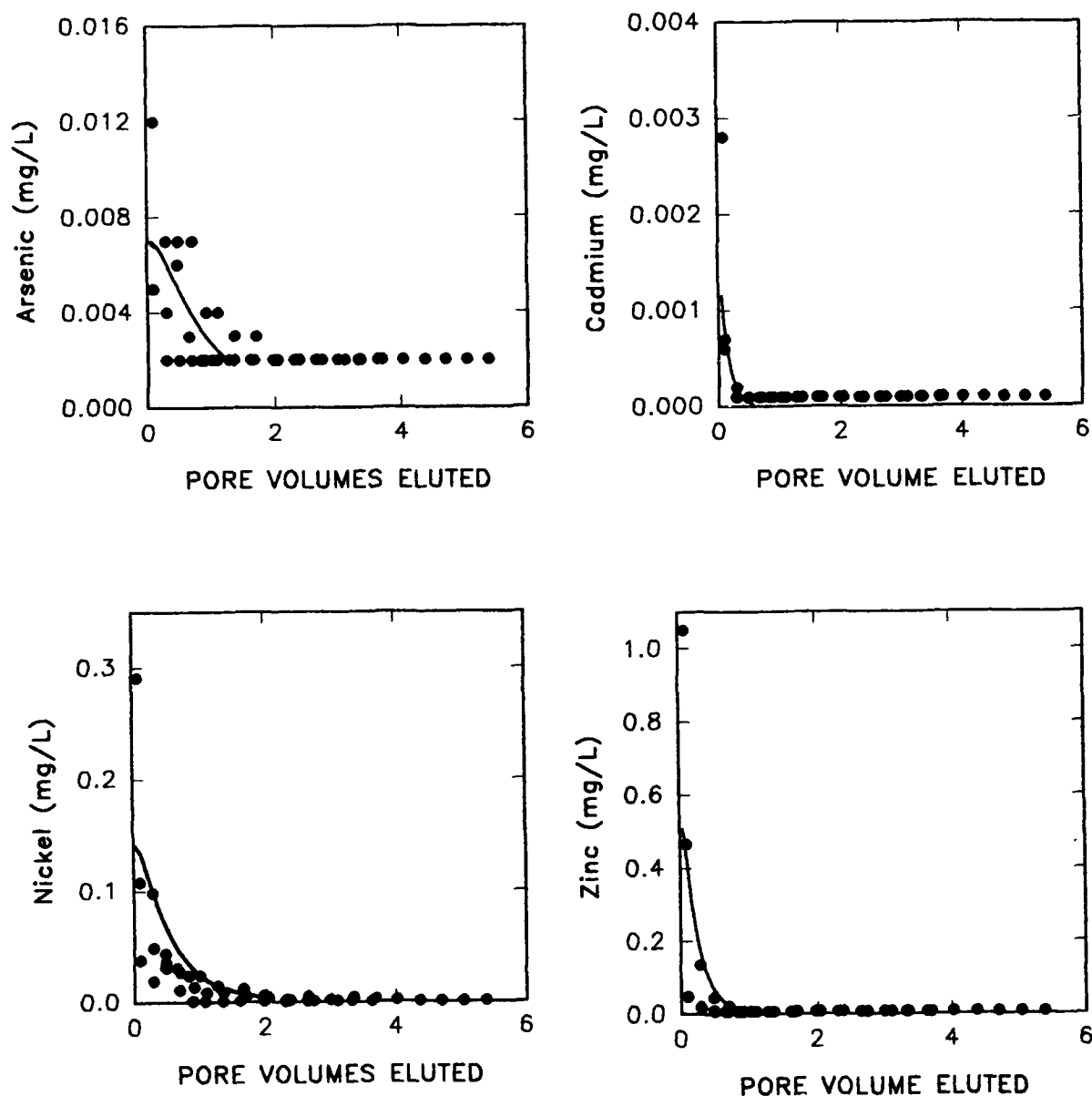


Figure 12. Predicted (line) and observed (dots) column elution data

discussed, analysis of the complete isotherm is not necessary. Using the first and second points of the zinc isotherm, the calculated distribution coefficient was 2.0 L/kg. Figure 12 shows the zinc elution curve predicted using this distribution coefficient and observed data.

Summary

74. Figure 12 indicates that Equation 11 provides a basis for modeling the leaching of As, Cd, Ni, and Pb from anaerobic Hamlet City Lake sediment.

A better fit for the model could be obtained by finding best-fit distribution coefficients, but the emphasis in this study was on testing the application of distribution coefficients obtained from sequential batch leach data. The integrated approach shows that batch studies provide important insights into the leaching of contaminants from dredged material, but may not always match results from continuous-flow transport experiments. Additional study is needed to understand the disparity between batch and column data for copper and to model the leaching of lead.

Leachate Impacts

75. Worst-case scenarios for anaerobic and aerobic leaching based on available batch and column data are discussed in this section. Separate anaerobic and aerobic scenarios are discussed as possible alternatives for Hamlet City Lake dredged material. For worst-case anaerobic leaching, maximum contaminant concentrations observed in anaerobic sequential batch leaching or column leaching are used, and for worst-case aerobic leaching, the maximum concentrations in aerobic sequential batch leachates are used. Worst-case leachate concentrations are compared to Federal and State of North Carolina criteria for ground and surface waters.

76. Neither hydraulic nor mechanical dredging adds sufficient oxygen to overcome the oxygen demand of dredged sediments. As a result, the dredged material in a CDF is anaerobic, except for a surface crust that slowly develops as the CDF dewateres by evaporation and seepage. CDFs containing fine-grained dredged material never completely dewater; therefore, a bottom layer that is saturated and anaerobic always exists. The oxidized crust may eventually become several feet thick, but generally fails to represent a significant portion of the vertical dredged material profile in fine-grained dredged material. Thus, leachate percolating into foundation soils will be anaerobic. Leachate quality in this case is simulated by anaerobic sequential batch and column leach tests.

77. Table 26 presents Federal drinking water limits and State of North Carolina groundwater standards for comparison to maximum anaerobic leachate concentrations. Maximum anaerobic leachate concentrations for As and Pb exceeded both standards. The maximum anaerobic Cd and Ni concentrations exceeded the State standard.

Table 26
Comparison of Leachate Concentration ($\mu\text{g/L}$)
to Regulatory Limits

<u>Constituent</u>	<u>Criterion</u>						
	<u>Max-AN*</u>	<u>Max-A**</u>	<u>DW†</u>	<u>AFC††</u>	<u>CFW‡</u>	<u>NC-GW‡‡</u>	<u>NC-SW§</u>
Arsenic	67	<2	50	360	190	50	50
Cadmium	5.4	9	10	3.9	1.1	5	2§§
Copper	164	<50	NC#	18	12	1,000	7
Lead	320	60	50	82	3.2	50	25
Mercury	<0.4	<0.2	2	2.4	0.012	1.1	1.2
Nickel	291	72	NC	1,800	96	150	88
Zinc	1,050	939	NC	320	47	5,000	50
Acenaphthene	<3	<1	NC	1,700	520	NC	NC
Fluoranthene	0.42	<0.1	NC	3,980	NC	NC	NC
Naphthalene	<2.4	<1	NC	2,300	620	NC	NC

* Maximum contaminant concentration in either batch or column anaerobic leachate, $\mu\text{g/L}$.

** Maximum contaminant concentration in aerobic batch leachate, $\mu\text{g/L}$.

† Level specified for compliance with Safe Drinking Water Act, $\mu\text{g/L}$.

†† Acute freshwater criteria, USEPA Water Quality Criteria for 1986, $\mu\text{g/L}$.

‡ Chronic freshwater criteria, USEPA Water Quality Criteria for 1985, $\mu\text{g/L}$.

‡‡ State of North Carolina Ground Water Standards, $\mu\text{g/L}$.

§ State of North Carolina Surface Water Standards, $\mu\text{g/L}$.

§§ Nontrout streams.

Data insufficient to develop criterion.

78. To fully oxidize dredged material, the material must be spread over a wide area in a thin lift so that evaporation and seepage can overcome the capillary forces that tend to maintain saturation. As water is removed, air penetrates the voids, and the oxygen demand of the sediment can be slowly eliminated. Eventually, the entire layer will be unsaturated and oxidized. Leachate quality in this case is simulated by aerobic sequential batch leach tests. For a thin lift of dredged material engineered and managed for dewatering and oxidation, aerobic leachate may seep laterally to surface waters. In this case, surface water criteria are applicable. If leachate flow is primarily vertical, groundwater criteria are applicable.

79. Federal acute and chronic freshwater criteria and State of North Carolina surface water standards are presented in Table 26 for comparison with aerobic leachate quality. Maximum concentrations in aerobic leachate exceeded

Federal acute and chronic water quality criteria and State of North Carolina surface water standards for Cd and Zn. Aerobic leachate also exceeded Federal and State of North Carolina groundwater standards for Pb and State of North Carolina standards for Cd.

80. Impacts on a receiving water, whether ground or surface, are determined not only by the concentration of contaminants in a discharge, but also by flow. The product of concentration and flow is mass loading. Mass loading maintained at high levels results in exposure concentrations that constitute significant environmental contamination. Flow for various disposal scenarios is discussed in the following section, although no criteria for contaminant mass loading exist against which comparisons can be made.

HELP Model Simulations

81. Leachate from dredged material placed in a disposal site is produced by three potential sources: the original pore water, rainfall infiltration, and, particularly for nearshore sites, groundwater or surface water in contact with the dredged material as a result of fluctuating water levels. For this analysis, the assumption was made that CDFs will be sited in an upland location where tidal pumping is not a factor.

82. Leachate generation and transport depend on site-specific hydrology, engineering controls at the disposal site, dredged material hydraulic conductivity, initial water content, and nature of contaminants. After dredging and disposal, dredged material is initially saturated (all voids are filled with water). As evaporation and seepage remove water from the voids, the amount of water stored and available for gravity drainage decreases. After some time, usually several years for conventional CDF designs, a quasi-equilibrium is reached in which water that seeps or evaporates is replenished by infiltration through the surface. The amount of water stored when a quasi-equilibrium is reached and the amount of water released before a quasi-equilibrium is reached are highly dependent on local hydrology, dredged material properties, and facility design factors. To predict time-varying leachate flow, all these factors must be considered.

83. Preproject estimation of leachate flow, therefore, requires coupled simulation of local weather patterns and hydrologic processes governing leachate generation. Important climatic processes and factors include precipitation, temperature, and humidity. Important hydrologic processes include

infiltration, snowmelt, runoff, and evaporation. Important subsurface processes include evaporation from dredged material voids and flow in unsaturated and saturated zones. The Hydrologic Evaluation of Landfill Performance (HELP) model (Schroeder et al. 1992) was used to simulate these processes for selected disposal scenarios.

84. HELP is a hydrologic water budget model that accounts for the effects of surface storage, runoff, infiltration, percolation, evapotranspiration, soil moisture storage, lateral drainage to leachate collection systems, and percolation through liners. Six alternative scenarios were selected to demonstrate use of the HELP model for estimation of percolation rates and to compare control measures. These scenarios are described below:

- a. Scenarios A and AL. Scenario A simulates a landfarm with a lift depth of 6 in. The volume of dredged material is 240,000 cu yd. No engineering controls are exercised for scenario A other than the routine operation and management needed to provide for drainage of surface runoff. Scenario AL is the same as scenario A except that the dredged material is placed on a 6-in. clay base.
- b. Scenarios B and BL. Scenarios B and BL are the same as scenarios A and AL except that dredged material is disposed in a single 3-ft lift. Confining dikes are needed for these scenarios.
- c. Scenarios C and CL. In scenarios C and CL, dredged material is placed in a single 10-ft lift inside a CDF. In scenario C, no engineering controls are exercised other than the routine operation and management needed to provide for drainage of surface runoff. In scenario CL, a leachate collection system and a composite liner are used to reduce the amount of leachate percolating through the bottom of the CDF. The liner consists of a 1-ft-thick barrier soil with a hydraulic conductivity of $1 \text{ E-}07 \text{ cm/sec}$ and a flexible membrane liner. Above the flexible membrane liner is a 1-ft layer of sand with a hydraulic conductivity of 0.01 cm/sec . Leachate collection pipes are placed every 600 ft in the sand layer, and drainage to the pipes is provided by a slope of 0.01 percent. The collected leachate could be treated by an onsite system or transported to an appropriate wastewater treatment facility.

85. HELP model runs for each of these scenarios used the same basic parameters (Table 27). Climatic data were generated for a 10-year period by the model, using default climatic data for Greensboro, NC. A Soil Conservation Service (SCS) runoff curve number of 88 was used. The surface of the dredged material was assumed to be devoid of vegetation, and the initial water content material was assumed to be the water content of the in situ sediment. Hydraulic conductivity of the dredged material was estimated to be $1 \text{ E-}05 \text{ cm/sec}$. This estimate is based on the grain size distribution and classification of the Unified Soil Classification System (USCS) (Figure 13)

Table 27
HELP Model Simulation Parameters

Dredged Material

Hydraulic conductivity -- 1 E-05 cm/sec
Porosity -- 0.81 cm³/cm³
Field capacity -- 0.222 cm³/cm³
Initial water content -- 0.81 cm³/cm³
SCS curve runoff number -- 88
Initial snow cover -- 0
Vegetative cover -- none

Clay Liner

Hydraulic conductivity -- 1 E-07 cm/sec
Porosity -- 0.43 cm³/cm³
Field capacity -- 0.366 cm³/cm³
Initial water content -- 0.43 cm³/cm³

Other

Flexible membrane liner leakage fraction -- 0.001
Climatic data -- Default data for Greensboro, NC

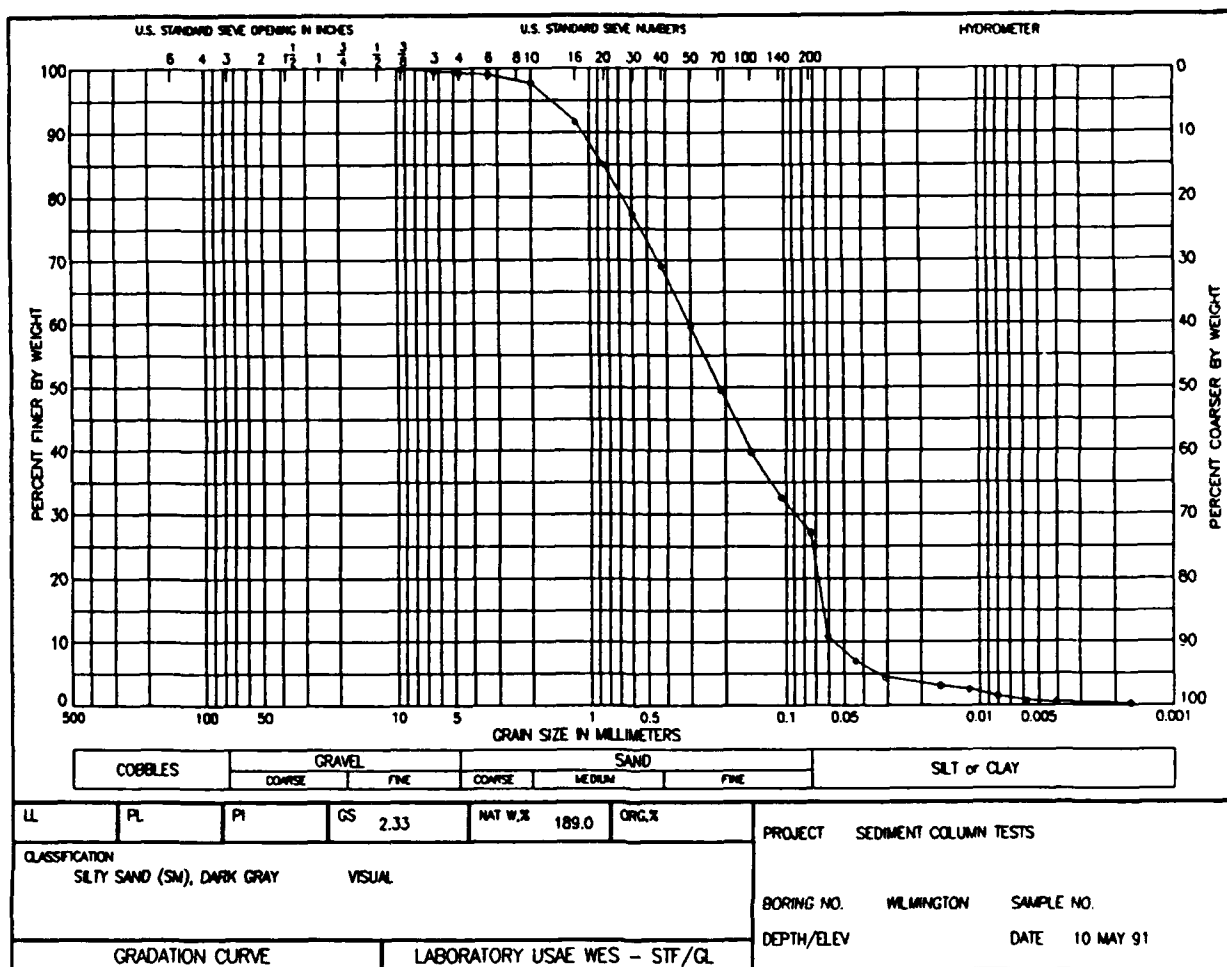


Figure 13. Grain size distribution for Hamlet City Lake sediment

and observations made during column leaching. Soils classified as SM under the USCS rarely have hydraulic conductivities less than $1 \text{ E-}05 \text{ cm/sec}$. During column leaching, pressure at the column entrance was less than 1 psig. For the flow and sediment column length used in the column leaching tests, a pressure gradient of less than 1 psig indicates that the hydraulic conductivity is $1 \text{ E-}06 \text{ cm/sec}$ or greater.

86. Figure 14 shows the annual leachate flow into foundation soils for the first 10 years of operation for scenarios A through CL. Table 28 lists the total leachate flow for the first 10 years for each scenario. The scenario with the thinnest lift and no controls (scenario A) has the highest percolation rate, and the alternative with a composite liner and a leachate collection system (scenario CL) has the lowest percolation rate.

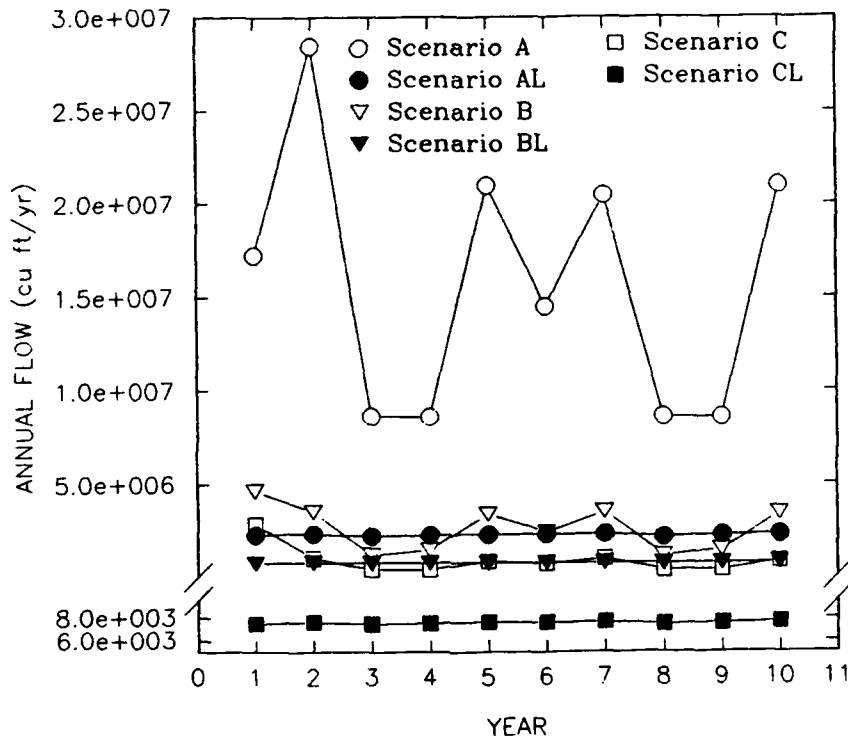


Figure 14. Annual leachate flow into foundation soils for HELP model simulations

Table 28
Cumulative Percolation Volumes from HELP
Model Simulations for First 10 Years

Scenario	Flow Into Soils Beneath Site cu ft
A	149,130,245
AL	22,880,615
B	26,388,375
BL	8,208,300
C	9,381,545
CL	75,335

87. Table 28 compares the summation of cumulative percolation volumes from the alternatives over the first 10-year period. Examination of results for tenth year volumes shows that, in terms of percolation into foundation soils, the scenarios can be ranked as follows: A > B > AL > C > BL > CL.

HELP runs for scenario A produced generally greater percolation volumes because of the increased surface area available for precipitation. Most of the reduction in percolation volume for scenario CL results from the drainage layer associated with the liner system. The HELP model indicates that the volume of total leachate collected in scenario CL by year 10 is 7.5 E+04 cu ft. To reduce overall contaminant losses from the disposal operation, this leachate should be treated.

PART V: CONCLUSIONS

88. Polycyclic aromatic hydrocarbons were found in low concentrations in Hamlet City Lake sediments. Only trace amounts of PAHs were found in leachate during batch and column testing. Total recoverable petroleum hydrocarbons were detected at relatively high concentrations in the anaerobic sediment. TRPHs were detected in the leachate during batch testing of anaerobic sediment only.

89. Results of this study showed that under disposal conditions similar to landfarming, organic contaminants such as the detected PAHs and TRPHs will decrease in concentration because of volatilization and/or biodegradation. Neither PAHs or TRPHs should therefore pose environmental problems. The data collected in this study were insufficient to estimate the rate of disappearance under field conditions; however, under selected laboratory conditions, more than 80 percent of the TRPHs disappeared in 6 months.

90. The pH of Hamlet City Lake sediment was slightly acidic (<6.0) while the sediment was anaerobic, and the pH was further decreased to less than 4.0 after oxidation. The data on metals leaching under disposal conditions similar to landfarming indicated that leachate quality is not seriously aggravated by dewatering and aeration of the sediment. Of the seven metals investigated in this study, three (arsenic, copper, and lead) were more mobile under anaerobic leaching conditions than under aerobic leaching conditions. Mercury did not leach under anaerobic or aerobic leaching conditions. For the three metals that were more mobile under aerobic leaching conditions (cadmium, nickel, and zinc), the mass release under aerobic batch leaching conditions was higher than that under anaerobic batch leaching conditions by a factor of only 2.5. A factor of only 2.5 does not indicate a substantial mobilization relative to the anaerobic condition.

91. Application of a contaminant transport equation and comparison of predicted and observed column elution data showed that anaerobic sequential batch leaching tests correctly predicted concentrations of PAHs and most metals in leachate under continuous-flow conditions. Therefore, critical aspects of the sequential batch leach procedure, such as shake time, liquid-to-solids ratio, and method of agitation, did not introduce artifacts that limit the utility of the data. Since aerobic and anaerobic sequential batch leach procedures are the same in these respects, the aerobic batch data should provide

a reliable indicator of probable leachate quality in sediments after dewatering and oxidation.

92. The data indicated that concentrations of only two metals (cadmium and lead) may exceed surface and groundwater criteria under the aerobic leaching conditions of a disposal scenario similar to landfarming. For anaerobic leaching conditions in a confined disposal facility, arsenic, cadmium, lead, and nickel may exceed groundwater criteria.

93. Simulations conducted with the Hydrologic Evaluation of Landfill Performance computer model showed that the leachate flow for a disposal scenario similar to landfarming was approximately 16 times the flow from a confined disposal facility (10-ft lift) with no liner. The distances and flow paths to receiving surface and ground waters, sorption properties of the disposal site soils, and local groundwater hydraulics are critical to full evaluation of differences in contaminant loading due to differences in flow. The absence of site-specific geohydrology further hinders proper interpretation of differences in flow.

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APPENDIX A: TOXICITY CHARACTERISTIC LEACHING PROCEDURE RESULTS

1. The Toxicity Characteristic Leach Procedure (TCLP) was conducted on freshly obtained Hamlet City Lake sediments provided by the Wilmington District. Sediments were obtained from the same general locations as were leachate samples.

2. The TCLP extracts were prepared using Method 1311 and were analyzed by the Analytical Laboratory Group of the U.S. Army Engineer Waterways Experiment Station using the following methods for metals:

<u>Analyte</u>	<u>EPA Method</u>
As	7060
Ba	6010
Cd	6010
Cr	6010
Pb	7421
Hg	7470
Se	7740
Ag	7761

Method blanks were run with the samples as well as external Quality Assurance samples of known concentration. Metals analyzed by Method 6010 were run on a Zeeman Labs Plasma Spec 3 sequential/simultaneous Inductively Coupled Argon Plasma Emission Spectrometer. The metals, As, Pb, Se, and Ag were run using a Perkin-Elmer 5100 Atomic Absorption (AA) Spectrometer with Zeeman background correction. Mercury was analyzed using a Perkin-Elmer 5000 AA.

3. Pesticides were analyzed using Method 8080, and herbicides were analyzed by Method 8150 using a Hewlett-Packard 5880A gas chromatograph with dual capillary columns and dual detectors. Quality control procedures as specified in the methods were followed.

4. Semivolatiles were analyzed by Method 8270 using an upgraded Hewlett-Packard Model 5985 GC/MS/DS with a wide-bore capillary column. All quality control procedures specified in the Method were followed.

5. TCLP concentrations for Hamlet City Lake sediment and the Regulatory Limit from the Federal Register (Vol 55, No. 61, 29 March 1990) are summarized in Table A1. As indicated in the table, all Hamlet City TCLP concentrations were below the Regulatory Limit.

6. The organics comprising the volatiles fraction (benzene, carbon tetrachloride, chlorobenzene, chloroform, 1,2-dichloroethane, 1,1-dichloroethylene, methyl ethyl ketone, tetrachloroethylene, and vinyl chloride) were

not subjected to TCLP analysis. Previous testing of Hamlet City Lake sediment at the WES did not indicate the presence of these constituents.

Table A1
Summary of Hamlet City Lake TCLP Results*

<u>Constituent</u>	<u>TCLP Concentration</u>	<u>Regulatory Limit</u>
Arsenic	0.016	5.0
Barium	1.05	100.0
Cadmium	<0.004	1.0
Chlordane	<0.0007	0.03
Chromium	<0.009	5.0
o-Cresol (2-Methylphenol)	<0.05	200.0
m-Cresol	<0.05	200.0
p-Cresol (4-Methylphenol)	<0.05	200.0
2,4-D	<0.012	10.0
1,4-Dichlorobenzene	<0.05	7.5
2,4-Dinitrotoluene	<0.05	0.13
Endrin	<0.00006	0.02
Heptachlor (and its hydroxide)	0.000017	0.008
Hexachlorobenzene	<0.05	0.13
Hexachloro-1,3-butadiene	<0.05	0.5
Hexachloroethane	<0.05	3.0
Lead	0.73	5.0
Lindane (G-BHC)	<0.00004	0.4
Mercury	<0.0002	0.2
Methoxychlor	<0.0018	10.0
Nitrobenzene	<0.05	2.0
Pentachlorophenol	<0.25	100.0
Pyridine	<0.05	5.0
Selenium	<0.002	1.0
Silver	<0.001	5.0
Toxaphene	<0.0025	0.5
2,4,5-Trichlorophenol	<0.05	400.0
2,4,6-Trichlorophenol	<0.05	2.0
2,4,5-TP (Silvex)	0.000081	1.0

* All concentrations are expressed in milligrams per liter.

APPENDIX B: DISPERSION COEFFICIENT MEASUREMENT

1. During the transit of water from the entrance of a column of sediment to the exit, water takes many different paths. Longer paths usually require longer travel times. The velocity at which water moves along the various paths varies because of differences in pore surface roughness and diameter. Thus, parcels of water introduced simultaneously at the entrance of the column can arrive at the exit at different times. The leachate that is collected at the exit, therefore, is a mix of parcels of water with different residence times in the column. This type of mixing is referred to as mechanical dispersion (Freeze and Cherry 1979).*

2. Molecular diffusion is another process affecting the quality of leachate exiting sediment columns. Together, molecular diffusion and mechanical dispersion are referred to as hydrodynamic dispersion. The significance of hydrodynamic dispersion can be determined by introducing a conservative tracer at the inflow and measuring the tracer concentration in column effluent.

Methods

3. After all leachate samples for metals analysis were collected from column 2, 2.125 ml of a 2.5-g/ml sodium chloride solution was added to the entrance tubing just below the entrance of the column. Distilled-deionized water was then pumped through the column at the flow used previously during the leaching study conducted with column 2. Leachate from the column was periodically collected and analyzed for electrical conductivity until the baseline electrical conductivity was reached.

Data Analysis

Data reduction

4. Electrical conductivity (EC) versus pore volumes eluted is shown in Figure B1. This figure shows a sharp leading front, followed by a long tail.

* See References at the end of the main text.

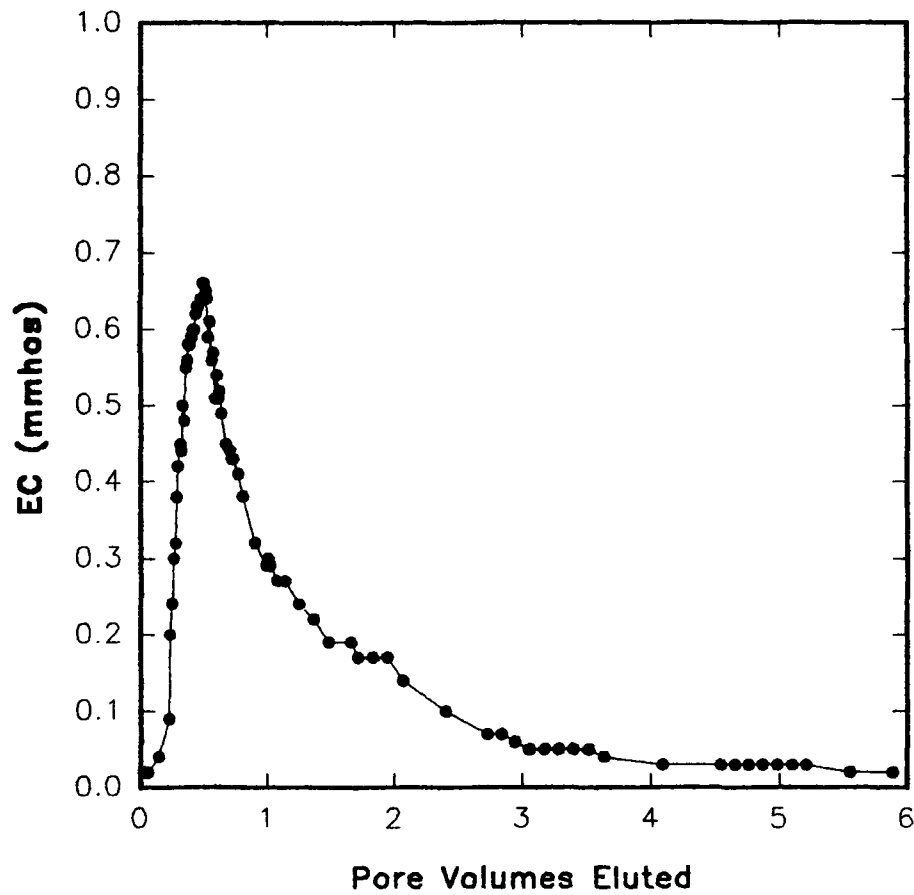


Figure B1. Electrical conductivity as a function of pore volumes eluted

The rapid rise in EC is the result of increased pore water velocities (pore water velocities higher than the average pore water velocity) in some channels. The long tail is the result of slow diffusion into and out of immobile water regions.

5. Methods described in Levenspiel (1972) were used to reduce tracer data. The variance of the EC curve was calculated as follows:

$$\sigma^2 = \frac{\sum_{i=1}^n (T_i)^2 E_i \Delta t}{\sum_{i=1}^n E_i T_i} - (\bar{T})^2 \quad (B1)$$

$$\bar{T} = \frac{\sum_1^n T_i E_i \Delta T}{\sum_1^n E_i T_i} \quad (B2)$$

where

σ^2 - variance, dimensionless

n - number of electrical conductivity measurements

T - pore volume eluted, dimensionless

i - index

E - electrical conductivity, millisiemens

6. The variance, σ^2 , is a measurement of the amount of spread or dispersion in the tracer elution curve. The greater the variance, the more hydrodynamic dispersion affects leachate quality at the column outlet. For coupling experimental curves with the mathematical concepts used in tracer work, it is particularly useful to normalize the variance as follows:

$$\sigma_\theta^2 = \frac{\sigma^2}{(\bar{T})^2} \quad (B3)$$

where σ_θ is the normalized variance (dimensionless).

7. The normalized variance obtained by the above calculation procedures was 0.913. For closed vessels, the normalized variance is related to dispersivity D as follows (Levenspiel 1972):

$$\sigma_\theta^2 = 2D - 2D^2 \left[1 - \exp \left(-\frac{1}{D} \right) \right] \quad (B4)$$

Equation B4 was solved using the secant method (Burden and Fairies 1978) to yield $D = 3.58$.

Interpretation and application

8. An intermediate amount of dispersion is indicated by a dispersivity value of 0.025, and a large amount of dispersion is indicated by a dispersivity value greater than 0.2 (Levenspiel 1972). The tailing shown in Figure B1

is responsible for the dispersivity value being as high as it is. As previously discussed, the tailing effect is the result of tracer diffusing into regions of immobile water (water sorbed to sediment solids and water trapped in dead-end pores).

9. During leaching experiments, some water will exit the columns after a relatively short travel time (less than 0.5 pore volume eluted) and, as a result, will tend to dilute the leachate collected at the column exit. In addition, contaminants will diffuse from immobile water regions into the pores in which water is flowing until the reservoir of contaminants in the immobile water region is depleted. The dispersion coefficient D_p is used to model these effects in the contaminant transport equation discussed as Equation 6 of the main text. Dispersivity and the dispersion coefficient are related as follows:

$$D = \frac{D_p}{VL} \quad (B5)$$

where

D_p - dispersion coefficient, cm^2/sec

V - average pore water velocity, cm/sec

L - sediment column length, cm

10. The dispersivity value obtained from the tracer study was used to calculate dispersion coefficients listed in Table 19 of the main text. The coefficients vary slightly depending on average pore water velocity.

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